STREAM COMMUNITIES IN ACID MINE DRAINAGES: INVESTIGATING LONGITUDINAL RECOVERY AND REMEDIATION POSSIBILITIES

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THESIS

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Abstract

The mechanisms of acid mine drainage were investigated in stream communities on the West Coast of the South Island of New Zealand, with specific regard to the Denniston/Stockton areas. To do this, a field survey of AMD impacted streams indicated that some natural recovery occurred without anthropogenic management, particularly in Charming Creek and Darcy Stream, and to a lesser extent in the Waimangaroa River. This recovery is associated with improvements in water quality due to the dilution of AMD by non-impacted tributaries. Rapid Creek is a small, contaminated stream that was used as a model to assess the mechanisms of AMD toxicity. Iron precipitate on substrate altered water chemistry and proved toxic to *Deleatidium* mayflies. Substrate with iron precipitate harbours more organic biofilm, but in-situ experimentation did not detect a difference in the invertebrate community associated with either iron-precipitate coated substrate, or substrate without iron. Experimentation over 96-hours indicated that low pH (<4) was the primary cause of toxicity, and that dissolved metals were secondarily toxic. Techniques to rectify these causes of toxicity were investigated. Dilution with distilled water, and the commercial cation-exchange products Bauxsol™ and Zeolite were used to treat AMD water, and the survival of *Deleatidium* subsequently exposed to this water was compared. Bauxsol™ and Zeolite improved mayfly survival after 96 hours by 50%. Dilution, up to 8-fold did not improve mayfly survival. Extrapolation indicated that dilution up to 64-fold would be required to mitigate the effects of acidic AMD entering Rapid Creek, even though the natural levels of pH are generally <5. The combined results of this study suggest that active remediation of AMD impacted streams would facilitate an improvement in the condition of the ecosystems, provided specific regard is given to the particular circumstances of each stream.

Chapter One

General Introduction

The extraction of coal has been a significant industry in the Buller region on the West Coast of the South Island since the 19th Century, and was fundamental to the establishment of the economy and communities there, and is still important to New Zealand's economy. In 1998 3.3 million tonnes of coal were exported, primarily to Japan, much of which was extracted from mines on the West Coast of the South Island (New Zealand Official Yearbook, 2000).

The abundant coal was first discovered near Charleston in 1846 by Charles Heaphy and Thomas Brunner (Coal Town Museum, 2001). In the 1860s Dr Julius von Haast, a geologist, and James Burnett, a colliery engineer, investigated the Denniston/Stockton Plateau. There they discovered substantial quantities of high quality coal (Coal Town Museum, 2001). In 1874 the first mine in the Denniston area established (Todd, 1989), and many more followed.

The government soon began to actively encourage mining to provide an exportable commodity and support the fledgling economy (Coal Town Museum, 2001). This saw a rapid increase in mining activities, and many communities soon grew in the Buller area. These included the townships of Mokihinui, Millerton, Granity, Ngakawau, Denniston, Coalbrookdale and Burnetts Face. Initially coal mines were primarily underground, but as infrastructure improved and technology advanced, other methods were introduced, including open pit mining and hydraulic sluice mining.

Gold was also discovered on the Denniston Plateau and in notable alluvial deposits in the Waimangaroa and Mokihinui Rivers. In the early 20th Century alluvial gold prospecting was becoming increasingly common. However, gold was not present in the same quantities as coal, and coal mining remained the primary economic activity in the Buller region for over 100 years (Coal Town Museum, 2001).

This rapid development of the mining industry and the unsustainable nature of mining activity have resulted in there being a large number of abandoned mines in the

Buller District. Although coal mining is still a significant part of the West Coast economy today, its importance has decreased markedly, particularly as modern mining methods focus on the use of machinery to extract coal in a much more labour-efficient process. This decline in importance has been mirrored by a decline in population in the Westport and northern West Coast areas (New Zealand Official Yearbook, 2000).

Both abandoned and active coal mines can have significant environmental impacts. The geology of the Brunner Coal Measures is composed of significant quantities of carbonaceous mudstone, which contains high concentrations of pyrite (Edguardo, 1997).

Pyrite is a sulphide-bearing mineral, which oxides when exposed to air and water. When this process occurs naturally it is referred to as acid rock drainage (ARD) and is generally gradual enough to have little effect on the ecosystems it enters (D.Bell, pers com). However, the process of mining accelerates the process, and results in a high level of acid mine drainage (AMD) (Campbell et al., 2000). This contaminates ground and surface waters and, as a consequence of seepage and runoff, profoundly degrades local waterways.

The sulphuric acid produced by the processes involved lowers the pH of streams. In addition to this, AMD water is also frequently associated with high levels of dissolved metals. In the Buller area, AMD impacted streams typically have elevated concentrations of iron and aluminium (Collier et al., 1990). These metals may also reach biologically toxic levels in stream waters, and in addition often form precipitates which alter the physical environment of streams (Harding et al., 2000).

This investigation focuses on the macroinvertebrates comprising the benthic community in order to study the effects of AMD on the ecology of impacted streams. In an investigation of streams in Westland, New Zealand Winterbourn (1998) found that the majority of streams studied that were influenced by coal mine drainage had pH levels below 3.6, and the fauna only consisted of a few insect species. Crustaceans, molluscs, platyhelminthes and annelids were all absent. The insect fauna that is likely to be found in mine drainages is consistently low in species diversity, low in abundance and dominated by Chironomidae (Winterbourn et al., 2000).

Fish are almost invariably absent from AMD impacted streams, as they are generally very sensitive to low levels of pH. In freshwater habitats where the ambient pH is less than 5 or 6, most fish are negatively affected. Acidification causes osmotic stress, impacts on the reproduction and recruitment success, and causes gill lesions (Herrman et al., 1993). Fish will eventually disappear from waters with persistently low pH levels (Fleischer et al., 1993).

Algae and microbes are also influenced by acid mine drainages. *Ulothrix,* a filamentous alga known to be highly acid tolerant, is frequently present at sites that are severely degraded by acid mine effluent (Winterbourn et al., 2000). Nordstrom (2000) found an array of green algae, bacteria, yeasts and fungi growing in water polluted by mine effluent. Microbial and algal communities are a good example of acidification and water chemistry driving succession and possibly even evolution on a localised scale (Herrman et al., 1993).

Because invertebrates are generally ubiquitous in freshwater habitats, invertebrate taxa are frequently used as indicators of the health of the aquatic system (bioindicators)(Collier et al., 1998). Invertebrate densities and diversities are relatively easy to monitor, and as they accumulate metals at similar concentrations to the substrate and water, they offer a quantitative approximation of the level of contamination of a particular habitat (Nelson & Roline, 1999).

Invertebrates are considered appropriate bioindicator agents for several reasons. Firstly, they are generally abundant and many taxa can accumulate metals in proportion to concentrations in the environment, whilst still tolerating them. Many invertebrate taxa are benthic and closely reflect concentrations of metals in the sediments. Most aquatic invertebrates are also relatively sedentary and have a short lifespan. Therefore, temporal and spatial fluctuations in habitat pollution are less likely to confound results. Invertebrates generally occupy lower trophic levels, so metal contamination assessments are unlikely to be falsely elevated by food chain biomagnification. Finally, the majority of benthic invertebrates in AMD impacted streams are generally immature insects, and so are not affected by reproductive processes such as gamete production (Goodyear & McNeill, 1999).

Many different techniques exist to utilise freshwater invertebrates for biomonitoring. These include taxon richness, Ephemeroptera-Plecoptera-Trichoptera (EPT) richness, chironomid richness, comparisons of dominant taxa proportions, and invertebrate density (Poulton et al., 1995). Although metal tolerances are highly variable between individual species (Goodyear & McNeill, 1999), the abundance of higher taxonomic groups is a good general indicator of metal pollution (Clements et al., 2000). However, the structure of an aquatic community with respect to the distribution of biological traits (e.g. mobility, recruitment success) is a better indicator of the effects of pollution than the use of species abundance, as it is not confounded by local community composition or spatial heterogeneity (Doledec et al., 1999).

Invertebrates are useful tools for monitoring the effects of pollution, but this practice has some limitations. For example, techniques are generally derived for specific situations and pollutants, and are often not applicable to other areas (Garcia-Criado et al., 1999). In addition, few studies have investigated the same organism, so little replicated data is available on the responses of specific taxa to specific metals (Goodyear & McNeill, 1999). However, 'sentinel' species can be identified which allow a sensitive assessment of the quantity of a pollutant that is biologically available, provided the sentinel species can be calibrated against the input of pollutants (Beeby, 2001).

AMD affects macroinvertebrates in several ways. Low pH can be directly toxic, as can high concentrations of metals. Acid mine effluent can contain high concentrations of ferric and ferrous iron, copper, zinc, lead, aluminium and magnesium (Parsons, 1977). In low pH conditions, iron can accumulate in the midguts of invertebrates, and may reduce or prevent the uptake of food (Herrman et al., 1993). The silt in a stream can become solidified by iron-flocculation, reducing the availability of interstitial spaces for refugia and habitat (Harding et al., 2000; Parsons, 1977). Iron precipitate can cover the gills of some invertebrates and impede oxygen uptake from the water (Harding et al., 2000). Furthermore, the chemical oxygen demand of acid mine effluent can result in streams having extremely low levels of dissolved oxygen, creating further difficulties for invertebrate gas exchange (Parsons, 1977).

Introduction to Rapid Creek

Rapid Creek is an example of an AMD impacted stream in the Buller District. It illustrates many of the characteristics of AMD contamination, including very low pH and elevated metal concentrations. It was therefore used as a model AMD impacted catchment in many facets of this study.

Rapid Creek is a small stream that flows off the western edge of the Denniston Plateau. It enters the Tasman Sea through a wetland area several kilometres south of the Waimangaroa River (Figure 1.1). Its total length is approximately eight kilometres and it is a third order stream when it reaches the ocean. The majority of the streambed of Rapid Creek is steep, bouldery and relatively unstable. The high annual rainfall (between 2000-8000 ml per annum) (West Coast Regional Council, 2003) of the West Coast Region, and the steep topography of the plateau margin combine to cause frequent high-flow and flood events in Rapid Creek.

The geology and vegetation of the catchment of Rapid Creek causes the surface water in the area to have a naturally low pH, which is generally around 4.5.

Rapid Creek is impacted by AMD in its upper catchment by Sullivans Mine (Figure 1.2).

Sullivans Mine was an extensive underground coal mine on the Denniston Plateau. It borders Rapid Creek on its Western (lower) boundary. Although coal was initially extracted from Sullivans Mine with traditional underground methods, later in the 20th Century hydraulic sluice mining was introduced as a faster and more efficient method of removing large quantities of coal (Coal Town Museum, 2001). This involved using high-pressure water to sluice coal out of the ground into the Rapid Creek streambed. From here it was flumed into large bins.

Although mining activities at Sullivans Mine have ceased and the adits in the Rapid Creek catchment are sealed, there is continual AMD water seepage from the mine. This water is very low in pH and has high concentrations of aluminium and iron. Furthermore high levels of arsenic and zinc have accumulated in the sediment due to extensive deposition during the use of hydro-sluice mining methods.

AMD has dramatically affected the ecosystem of Rapid Creek, and reduced the aesthetic appeal of the area. The substrate is stained orange by iron-precipitate (Figure 1.3) and the benthic community is low in species richness and abundance. Fish are entirely excluded from Rapid Creek by the low pH.

Because Rapid Creek is easily accessed and therefore visible to the public, it has been identified as an appropriate candidate for restoration efforts. By using it as a case study, it is intended that useful techniques for AMD remediation in the Greymouth/Westport/Buller areas can be learned. These techniques may be later applied to other AMD affected catchments like Soldiers Creek and Ford Creek near Blackball, Seven Mile Creek north of Greymouth, and several streams in the Stockton and Millerton area.

Figure 1.1 Rapid Creek, upstream of Sullivans Mine. Photograph courtesy of J. Harding.

Figure 1.2 Water draining from a sealed adit of Sullivans Mine. Photograph courtesy of J. Harding.

Figure 1.3 **Rapid Creek, downstream of AMD from Sullivans Mine. Iron precipitate on substrate is clearly visible. Photograph courtesy of J. Harding.**

General Objectives

This study investigates the mechanisms of acid mine drainage and focuses on the unique climatic, topographic and physicochemical situation on the West Coast of the South Island of New Zealand.

A field survey of larger AMD impacted catchments was used to assess if recovery occurred without anthropogenic management, particularly in respect to the longitudinal distance downstream from the source of AMD (Chapter Two).

This study also explored the mechanisms of AMD toxicity (Chapter Three) using the physicochemical conditions of Rapid Creek as a model. The affects of iron precipitate on substrate were investigated, including the direct toxicity of the precipitate and the possibility that it dictates habitat selection in macroinvertebrates

(Chapters Three and Four). Additionally, experimental work evaluated several potential remediation techniques and products, again with regard to the water chemistry and existing ecosystems of Rapid Creek specifically, with the intention of extrapolate to other Buller District catchments (Chapter Five).

The overall objective was to consolidate this information on natural recovery, mechanisms of toxicity and remediation possibilities to produce a conceptual model that illustrates natural and mediated improvement in water chemistry and habitat conditions, and the resulting recovery of stream ecosystems from AMD impacts (Chapter Six). This will enable the construction of an ecological foundation for future restoration and remediation of AMD impacted streams on the Denniston Plateau in the Buller District, and potentially throughout the West Coast Region.

Chapter Two

Longitudinal survey of the degradation and recovery of macroinvertebrate communities in AMD impacted catchments

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Extensive mineral extraction operations in the Buller District, both past and present, have resulted in long-term impacts on water quality and sediment levels in many catchments (Harbrow, 2001). This is reflected by changes in the composition of benthic macroinvertebrates communities in many of these mine drainages (Winterbourn, 1998). Acid mine drainage (AMD) into streams generally causes a reduction in pH and frequently an elevation of dissolved metals like iron and aluminium in the water and in the substrate. This affect is particularly pronounced in the Stockton-Denniston Plateau area, where streams collect drainages from mines in the Brunner Coal measures. These coal measures are particularly high in sulphur, and are also frequently associated with iron pyrite nodules (Edguardo, 1997). When exposed to water and air, these minerals oxidise, resulting in many waterways having a very low pH and high concentrations of dissolved iron (Alarcon Leon & Anstiss, 2002; Campbell et al., 2000; Webster-Brown & Brown, unpublished data). Numerous other minerals are also associated with AMD from the Brunner Coal Measures. These include toxic metals like aluminium, arsenic, zinc and cadmium, which are generally found at lower levels than iron, although they can still be present in sufficient concentrations to be toxic to invertebrates (Winterbourn, 1998).

The impacts of AMD have been widely documented, and systems affected by mine inputs typically have benthic communities that are low in taxonomic diversity (Rosemond et al., 1992; Sutcliffe & Hildrew, 1989). Allard and Moreau (1986) found that by simulating AMD conditions in stream channels (reducing pH and adding aluminium), the benthic macroinvertebrate communities showed a marked reduction in taxa richness, and became dominated by Chironomidae species. The biomass of invertebrates remained the same in these 'impacted' channels, as the chironomids increased in size and number as they were released from feeding competition and predation pressure as more sensitive species succumbed to the toxicity of the conditions (Allard & Moreau, 1986).

In central and northern West Coast streams pH is strongly correlated with iron and aluminium concentrations (Harbrow, 2001), and with macroinvertebrate diversity (Anthony, 1999), so it is likely that sensitive taxa are restricted by a combination of pH and metal toxicity, as well as physical changes to substrate and habitat quality. These West Coast streams typically show a pattern of decreased taxonomic richness and increased dominance of pollution tolerant taxa (Harbrow, 2001).

The presence of excessive iron flocculation in these streams is a further limiting factor for macroinvertebrates, as it consolidates the substrate and reduces habitat heterogeneity (Harding et al., 2000; Penny, 1987). This degradation of the benthic habitat by clogging interstitial spaces and reducing the refugia available to invertebrates (Harding et al., 2000) may be particularly crucial in stream environments, such as these, that are subject to acute flow fluctuations. The absence of interstitial refugia for invertebrates would make these animals very vulnerable to being dislodged. The physical degradation of habitat by mining activities is probably amplified by the high annual rainfall in the West Coast region (West Coast Regional Council, 2003). Rainfall is between 2000-8000mL per annum, which results in streams frequently being subjected to flood events.

The New Zealand fauna does show a degree of tolerance to these stresses. Moderately AMD-impacted streams (pH 3 –4.5) commonly have Plecoptera taxa present (notably *Zelandobius*), and *Oxyethira* (Trichoptera), a pollution-tolerant caddisfly (Harbrow, 2001), (pers obs) whereas *Deleatidium* (Ephemeroptera), a ubiquitous leptophlebiid mayfly is generally sensitive to contamination. However, *Deleatidium* has been found in several streams with low pH (>4) and high concentrations of iron and aluminium (>3 mg L^{-1}) (Winterbourn et al., 2000b). In more severely contaminated streams (pH<3) benthic communities are commonly dominated by a few chironomid taxa, including *Chironomous zealandicus*, and *Cricotopus sp.* (Harbrow, 2001). In water of pH less than 3, Ephemeroptera, Plecoptera and most Trichoptera are generally excluded (Winterbourn, 1998). Molluscs and crustaceans are noticeably absent from AMD impacted streams, due to a lack of calcium for the construction of carapaces and shells (Harding et al., 2000), and the fact that these animals tend to rely on seeking refugia from high-flow disturbance events (Holomuzki & Biggs, 2000).

New Zealand streams are by nature short in length, and subject to frequent flood events. This harsh environment has resulted in a lack of specialisation in freshwater macroinvertebrates, and in many streams having low taxonomic richness compared to longer, more stable streams in other countries (Winterbourn et al., 1981). Furthermore, high concentrations of humic substances in streams that flow through native forest frequently results in West Coast streams having a naturally low pH (Collier & Winterbourn, 1987). This is particularly common in areas of high rainfall, where increased leaching of vegetation occurs. Macroinvertebrates in these naturally acidic environments exhibit a degree of tolerance to low pH, and taxonomic richness is similar in these streams to that of circum neutral streams (Collier & Winterbourn, 1987). Although similar concentrations of aluminium may arise in naturally acidic streams, aluminium is considered to be less toxic due to a chemical reaction with dissolved organic carbon (Winterbourn & McDiffett, 1996).

These factors are particularly pronounced on the Stockton/Denniston Plateau where catchments are typically short and steep (often less than 10 km), rainfall is high, and native forest is dense and high in humic matter. Naturally acidic brown-water streams are common on the West Coast, and therefore form an important component of freshwater habitat (Collier et al., 1990). Acidic streams with a pH of around 4.5 are frequently found to have equally diverse benthic communities to more circum-neutral streams in the areas (Collier et al., 1990). Frequent flooding events are also likely to have fostered rapid recolonisation ability in macroinvertebrate populations (Winterbourn et al., 1981). This is likely assisted by the life histories of many taxa, as they are generally not dictated by season (Winterbourn et al., 1981), and can often produce many generations in a year.

Because West Coast stream fauna have a degree of adaptation to naturally acidic conditions, and have evolved in harsh physical environments, it may be possible that benthic macroinvertebrate communities can recover diversity downstream of impacted sites, once the initial acute toxicity of the AMD has been diluted by tributaries.

This study aimed to:

1. Investigate benthic macroinvertebrate communities longitudinally in mine impacted-streams to establish if "recovery" occurs. Recovery in AMD impacted benthic macroinvertebrate communities is likely to include a change in community composition from being low in taxonomic diversity,

dominated by a few pollution-tolerant taxa, to a more diverse assemblage, incorporating sensitive taxa.

2. To assess if community composition changes are associated with improvements in water chemistry and longitudinal distance, and therefore establish important factors in the natural recovery of AMD degraded stream ecosystems.

Community composition is expected to alter as a waterway increases in stream order, size and morphology (Vannote et al., 1980). Therefore, in order to identify recover in communities, reference catchments were also sampled to provide a baseline for normal streams and rivers in the area, and allow these expected longitudinal changes to be to factored into the analysis.

Study Area

A survey of seven catchments was carried out during February-March, and September-October, 2002 (Figure 2.1). Where possible, catchments were paired, one impacted catchment, affected by AMD, paired with one reference (control) catchment, that is free from AMD contamination. Reference catchments were selected based on similarities in size, elevation and stream order to impacted catchments. Stream systems impacted by other anthropogenic land uses (eg dairying) were rejected; both for selection as AMD impacted and control catchments.

In three cases paired catchments were sampled, with a mine impacted catchment paired with a 'reference' catchment, where no mining has occurred within the system (Table 2.1). Impacted catchments were affected by mining activities within the Brunner Coal Measures.

The Waimangaroa River catchment arises on the Denniston Plateau, approximately 500 metres above sea level, and flows westward through a deep gorge before joining the Tasman Sea at the township of Waimangaroa. The Waimangaroa River is approximately twelve kilometres long from its headwaters to its delta, by which stage it is a fourth-order river (Figure 2.3). This river receives AMD from several historic mines, most notably the Coalbrookdale and Burnett's Face workings. There are no operating mines currently discharging into the Waimangaroa River.

The majority of the length of the Waimangaroa River is run-riffle habitat, with cobble/boulder substrate. There are some pools, chutes and waterfalls where bedrock is exposed through the gorge. The riparian margin of the Waimangaroa is almost entirely vegetated with native forest, with the exception of the lower 3-4 kilometres, which flow through farmland.

The Little Wanganui River was used as a comparative (reference) catchment for the Waimangaroa River. The Little Wanganui River catchment is of a similar size to that of the Waimangaroa River, and arises in the Matiri Ranges. It flows westward through native forest and farmland to join the Tasman Sea approximately 15 kilometres south of Karamea.

The Little Wanganui River is longer than the Waimangaroa River (approximately 25 km compared to 15 km), so only the lower reaches (4^{th} and 5^{th} order) were surveyed for this study, although a minor tributary was used a comparison for the upper Waimangaroa sites (approximately 340 metres above sea level). The bed of the Little Wanganui River is dominated by boulders and large cobbles, although within three kilometers of entering the sea the river bed is composed of smaller cobbles and sand (Figure 2.2). The Little Wanganui River is dominated by riffle habitat in its upper reaches, whereas in the lower reaches the river is primarily run habitat.

Charming Creek flows through the Mokihinui Forest into the Ngakawau Gorge. The native forest in the area has been extensively milled in the past, and there is still some production forestry in the catchment. The Charming Creek Coal mine is abandoned, but continues to contaminate Charming Creek via a severely impacted tributary, Wearne Creek. Charming Creek is a smaller catchment than that of the Waimangaroa and Little Wanganui Rivers. The studied reach, from upstream of the mine to its confluence with the Ngakawau River is less than six kilometres. Charming Creek is relatively low lying, with most of its length lying below 150 metres above sea level. Charming Creek is unique in this study in that the majority of its length the stream bed is bedrock, although there are some areas of cobbles and gravels. Because Charming Creek has uniform substrate (bedrock), most of the stream is dominated by run habitat (Figure 2.4). However, in some areas where there are cobbles on the stream bed there is riffle habitat. The bedrock also creates some chutes and waterfalls in the bed of Charming Creek.

Tidal Creek, which arises in the Radiant Range, was used as a reference catchment for Charming Creek. It flows alongside State Highway 67 north of the Karamea Bluff. It mainly has a densely native forested catchment, although there is some farmland in the lower parts of the catchments. The surveyed reaches of Tidal Creek are low lying, similar to those of Charming Creek, with an altitude range of 40-120 metres above sea level. Tidal Creek has predominantly small and large cobbles, and the stream morphology is primarily riffles, although there are some runs and pools.

Miller Creek is a short waterway that starts above and around the settlement of Millerton. It flows through the settlement and joins Granity Stream and Mine Creek before flowing into the sea at Granity (Figure 2.5). There are many mines in this small catchment, but all are now abandoned. Although the area was cleared of vegetation, the cessation of mining has facilitated the regeneration of native forest. Although the steep topography of the Miller Creek catchment gives rise to several waterfalls, the majority of the stream is cobbly riffles with some large boulders.

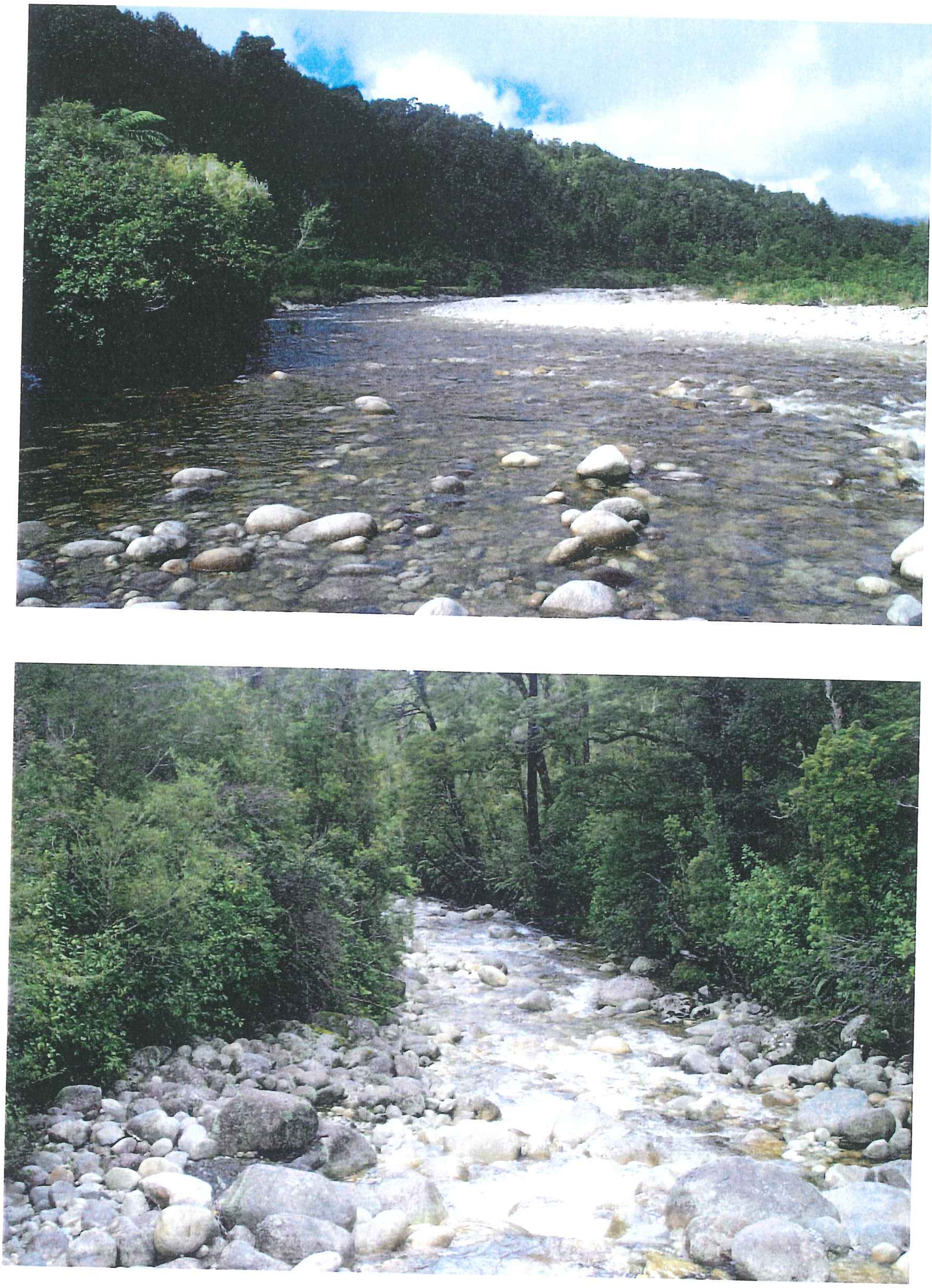
Chasm Creek was used as a reference catchment for Miller Creek (Figure 2.2). Its catchment is in the Mokihinui Forest, and it flows north into the Mokihinui River near Seddonville. The catchment is densely forested with both native and exotic forest. Chasm Creek also has predominantly riffle habitat, and the substrate consists of both large cobbles and boulders.

Darcy Stream was an AMD impacted stream flowing into the Ngakawau River from the Stockton Plateau. It is impacted by drainage from the abandoned Mount William Coal Mines. No direct catchment partner for Darcy Stream was surveyed, but it is similar to Miller Creek, and therefore Chasm Creek is a suitable comparison. Like Miller Creek, the Darcy Stream catchment is vegetated with regenerating native forest, with some patches of first generation bush (Figure 2.6). The substrate in Darcy Stream is a combination of cobbles, boulders and bedrock, with some areas of finer substrates. Darcy Stream morphology is diverse, including pools, runs, riffles, chutes and cascades.

Table 2.1 **Details of the sections of catchments investigated for the survey of longitudinal recovery from AMD impacts**

Catchment	AMD impacted or reference	Altitude Range (msl)	Stream Order Range	Associated Mines	Riparian Vegetation
Waimangaroa River	Impacted	0-540	2-4	Burnetts Face Coalbrookdale Others	Regenerating native forest
Little Wanganui River	Reference	0-340	4-5	-	Mature native forest
Charming Creek	Impacted	120-80	2-3	Charming Creek Mine	Mixed regenerating/mature native forest, exotic spp (gorse)
Tidal Creek	Reference	100-0	1-3	-	Mature native forest
Miller Creek	Impacted	400-0	2-3	Old Dip Coal Mine	Regenerating native forest
Chasm Creek	Reference	320-180	2-3	-	Mature native forest/exotic conifer forest
Darcy Stream	Impacted	640-400	1-4	Mt William Coal Mine	Mature native forest

Figure 2.1 **Location of the seven streams surveyed.**



(a)

(b)

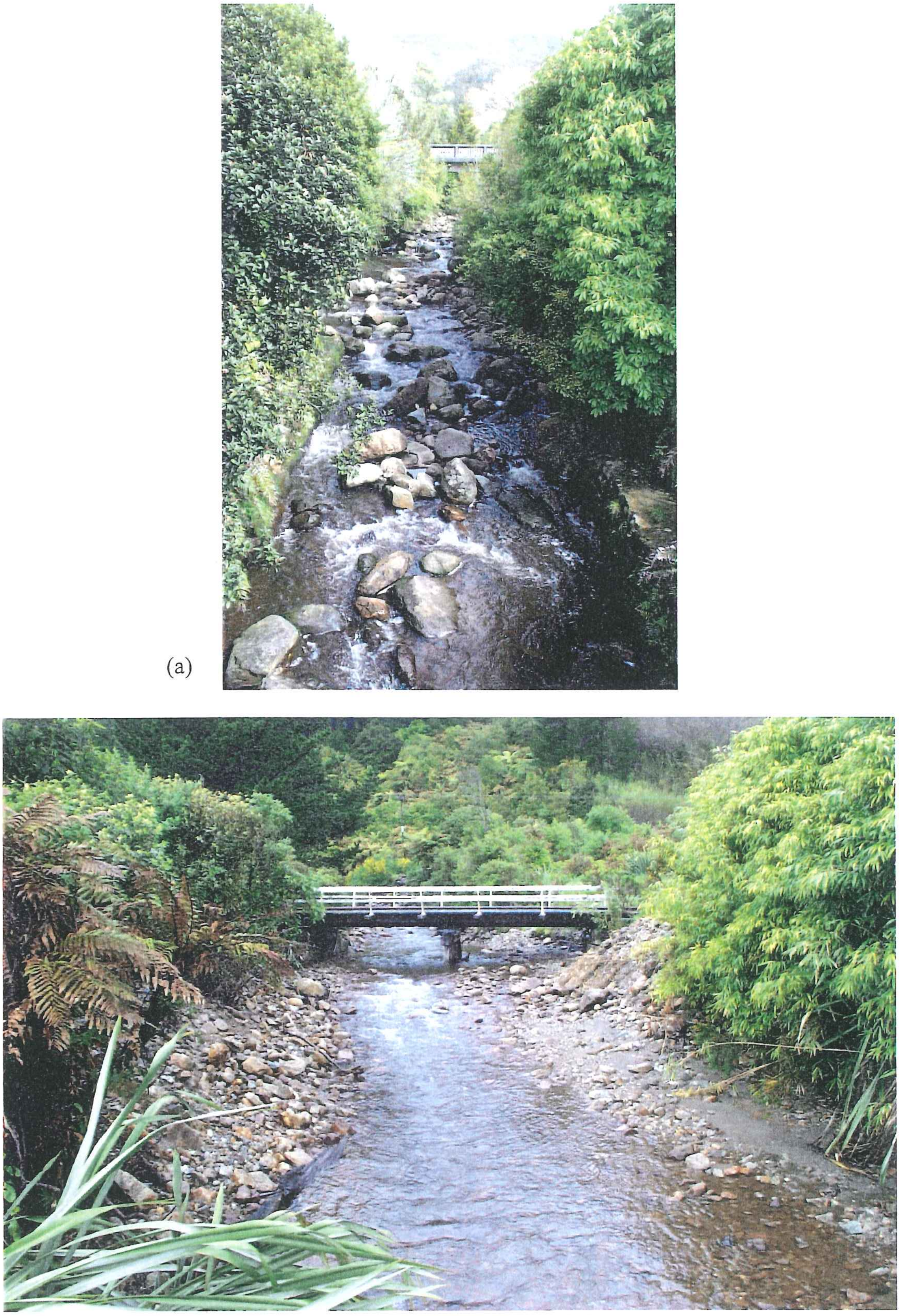
Figure 2.2 **Reference catchments: Little Wanganui River (a), and Chasm Creek (b).**

(a)

(b)

Figure 2.3 **Waimangaroa River, Site 2 above the gorge (a), and Site 3, near the Waimangaroa township (b).**

Figure 2.4 AMD impacted Charming Creek. Note iron stained bedrock, and non-impacted tributary entering on the true right (lower photo).



(a)

(b)

Figure 2.5 **Miller Creek. Upper reaches (a), and the lower reaches after confluence with Granity Stream (b).**

(a)

(b)

(c)

Figure 2.6 **Darcy Stream, upper reaches (a), lower reaches (b and c). Photographs courtesy of A.Black, CRL Energy.**

METHODS

Four or five study reaches (30 m long) were selected on each stream, depending on available access and longitudinal distance between the point of AMD impact and the lowest accessible site. In the AMD impacted catchments, one site was upstream of the mine outflow (when possible), one was immediately downstream of the mine outflow, and two-three were evenly spaced downstream of the mine outflow. The most downstream site was generally just prior to the confluence of the stream with a larger river, or the sea.

Sites in the control catchments were selected to represent similar elevation, stream order and longitudinal separation as equivalent sites in AMD impacted catchments.

A range of physico-chemical parameters were measured at each site; pH and conductivity were measured in-situ using an Oakton® Waterproof pH/Con 10 Meter, and dissolved oxygen was measured with a YSI Incorporated Model 57 Dissolved Oxygen meter. Both meters were appropriately calibrated before each use.

The concentration of sulphate ions was measured by collecting a water sample in an acid-washed polyethylene bottle, and analysed using a HACH DR2000 Direct Reading Spectrophotometer according to the method described in (Hach, 1992). Sulphate was measured because it is considered a good indicator of the magnitude of AMD influence on stream systems (J.Webster-Brown, pers comm). Although sulphate is non-toxic to aquatic life, in the Brunner Coal Measures it is associated with toxic trace metals and pH. It is faster and easier to measure than metals, and is not corrupted by other factors as pH is (organic acids, carbonate buffering) (Webster-Brown & Brown, unpublished data).

Width was calculated by averaging the length of three randomly selected transects within the study reach. Depth was measured at three random points and averaged. Surface velocity was estimated by timing how long it took a neutrally buoyant object (an orange) to be carried 20 metres by the current. This was repeated three times and an average velocity was obtained.

Streambed and substrate stability was determined using the Pfankuch channel stability method (Pfankuch, 1975). Altitude for each site was taken from NZMS 1:50 000 topographical maps.

The biological community was quantified at each site by collecting three Surber samples (0.1 m^2; 0.5mm mesh). Samples were collected from random positions within riffle habitat wherever possible. A further composite kick net sample was collected from all microhabitats present at each site. Samples were preserved in 70% ethanol and returned to the laboratory for processing.

Macroinvertebrate samples were sieved through a 500μm mesh, sorted, enumerated and identified. Individuals were identified to genus when possible, and otherwise to the lowest feasible taxonomic level (Winterbourn et al., 2000a), magnified up to 35x using a binocular dissecting microscope. Chironomidae were mounted on slides with lactophenol PVA and identified to genus by magnifying 100x – 400x using a stereo microscope (Winterbourn et al., 2000a; Taylor, 2001).

Periphyton / organic biofilm biomass was quantified collecting three replicate samples from each site. A specific area of stone surface was scrapped carefully with a nylon brush, and the periphyton removed collected in a small quantity of stream water (Davies & Gee, 1993). Samples were stored in the dark on ice, and returned to the laboratory for analysis.

Particulate matter was filtered out of the water using pre-ashed glass filters. These were then dried for 12 hours at 65 °C. The dry-weight of samples was then measured, before they were ashed for one hour at 450 °C. The ash-free dry mass of each sample was therefore established, and then averaged to give a mean mass of organic matter present per m^2 of substrate.

Statistical Analysis

Density data was obtained by averaging the three Surber samples from each site, then converting to density per m^2. Total taxa were established by combining all taxa from in Surber and kicknet samples. The percentage of the community that was comprised of Ephemeroptera, Plecoptera and Trichoptera individuals was established by dividing the number of EPT individuals by the total number of individuals found in a sample.

A series of two-tailed t-tests were used to assess the suitability of control catchments as partners for impacted catchments, by comparing the mean of individual physico-chemical variables for each catchment in a pair (Microsoft Excel).

Quantitative data was log transformed to improve normality, and then used in a Cluster Analysis to compare the macroinvertebrate communities of the longitudinal sites (McCune & Mefford, 1999). A Two-Way Indicator Species Analysis (TWINSPAN) was also performed to identify important species in both AMD impacted and control catchments (McCune & Mefford, 1999).

RESULTS

Physico-chemical

The wide range of values for physical and water chemistry data (Table 2.2) illustrates the variable nature of West Coast streams. The sites surveyed represented stream orders in ranging from 1st order to 5th order.

T-tests were performed on the physico-chemical data to establish which parameters were significantly different in AMD impacted streams than in reference streams. Table 2.3 shows that stream order, conductivity, width, velocity, and channel stability scores were not significantly different in the two catchment types surveyed. Sulphate, pH, depth and dissolved oxygen were all significantly different in the two catchment types. This suggests that the majority of physical features were independent of AMD impacts, as opposed to water chemistry parameters that were not. Conductivity is a notable exception, as it was not significantly different between AMD impacted catchments and reference catchments. However, the mean conductivity in AMD streams was 289.3 µS, as opposed to 88.2 µS in reference streams. It is likely that the high variation in conductivity data between sites is responsible for the inability to detect a statistical difference.

The AMD impacted streams had a lower concentration of dissolved oxygen. Mean dissolved oxygen in AMD impacted streams was 8.01 mgL^{-1}, whereas reference streams had a mean of 11.59 mgL-1 ($T=3.6_{28}$, $p=0.001$).

Sulphate and pH both indicated the presence of AMD influence, and were significantly different between AMD and reference catchments. Sulphate was high in AMD impacted catchments (80.25mgL^{-1}), and rarely within the detection range in reference catchments ($T=2.9_{28}$, $p=0.007$). Miller Creek had the highest concentration of sulphate, with > 300mgL^{-1}. Of the AMD catchments, the Waimangaroa River had the lowest concentrations of sulphate, with a maximum concentration of 20 mgL-1.

Similarly, pH was much lower in AMD impacted streams, with a mean of 4.9, as opposed to a mean pH of 6.8 in reference catchments ($T=-3.3_{28}$, $p=0.003$). Miller Creek and Darcy Stream had the lowest levels of pH. All AMD sites in these streams had pH levels less than 3.5. The Waimangaroa River and Charming Creek had more moderate pH levels, generally between 5 – 7. This is comparable with the reference catchments, particularly Chasm Creek, which has naturally acidic water, within a pH range of 5-6.

Table 2.2 Physico-chemical parameters of all sites surveyed in 2002.

Stream	Site Code	AMD impact ?	Longitudinal Distance (km)	Sulphates (mgL^{-1})	pH	Conductivity (μScm^{-1})	Temperature (°C)	D. O. (mgL^{-1})	Stream Order	Mean Width (m)	Mean Depth (m)	Mean Surface Velocity (m/s)	Channel Stability
Waimangaroa River River	W1	N	0	0	7.25	71.5	10.7	5.29	2	1.4	0.2	-	65
	W2	Y	1	20	4.98	70.3	15.3	5.80	3	8.7	0.2	0.35	65
	W3	Y	8.5	19	7.08	82.2	20.6	5.15	4	24.3	0.3	0.24	78
	W4	Y	9.5	19	6.44	125.4	22.1	11.59	4	30.2	0.6	0.42	77
Charming Creek	C1	N	0	1	6.77	85.2	19.5	7.44	2	4.8	0.3	0.32	56
	C2	Y	0.5	40	5.86	144.3	18.6	9.51	3	6.0	0.2	0.38	60
	C3	Y	2	36	6.41	133	19.7	6.06	3	5.3	0.2	0.71	60
	C4	Y	3	29	6.81	138.6	16.1	7.91	3	6.5	0.4	0.41	57
	C5	Y	4	29	7.0	126.5	16.5	7.26	3	4.8	0.3	0.48	49
Miller Creek	M1	Y	0	360	2.8	1942	12.0	9.5	2	1.0	0.3	-	78
	M2	Y	0.75	244	3.3	350	10.3	11.28	2	4.7	0.3	0.22	67
	M3	Y	2	168	3.2	544	11.0	11.18	3	3.0	0.7	-	92
	M4	Y	4	140	3.0	699	11.3	10.35	3	3.1	0.4	0.94	94
Darcy Stream	D1	N	0	5.6	4.3	39	7.5	6.7	1	1.6	0.3	-	55
	D2	Y	0.5	116	2.61	125	7.2	7.2	1	1.6	0.3	-	84
	D3	Y	2.5	93.9	2.81	125	9.4	7.1	2	2.3	0.5	-	84
	D4	Y	4	43.8	3.11	117	9.9	6.8	4	5.3	0.6	-	49
Little Wanganui River	LW1	N	0	0	7.53	65	14.2	8.3	4	2.7	0.4	0.70	99
	LW2	N	1	0	7.46	66.9	14.5	7.11	4	17.7	0.7	0.92	88
	LW3	N	4	0	7.45	68.2	16.2	7.7	4	15.7	0.6	0.90	69
	LW4	N	8.5	0	7.74	104.6	20.5	5.72	5	14.7	1.4	1.0	120
Tidal Creek	T1	N	0	0	6.55	74.3	13.6	14.04	1	1.9	0.4	-	56
	T2	N	0.7	0	7.83	158.6	13.5	14.55	3	3.4	0.3	0.6	46
	T3	N	1.7	0	7.79	159.3	13.0	15	3	5.6	0.4	1.14	50
	T4	N	3.2	0	7.58	139.4	12.6	14.76	3	6.1	0.3	0.93	49
	T5	N	4	0	7.05	102.7	14.4	14.99	3	4.9	0.7	0.50	48
Chasm Creek	CS1	N	0	1	5.3	51.1	7.8	12.34	2	4.7	0.5	1.20	53
	CS2	N	1	1	5.2	37.9	7.7	11.63	2	8.7	0.60	0.60	56
	CS3	N	3	0	5.4	58.9	8.8	11.89	3	5.6	0.7	0.42	79
	CS4	N	4	0	5.6	59.1	8.9	12.6	3	13.3	0.62	0.39	58

Table 2.3 Comparison of the mean physico-chemical characteristics (±SE) of AMD impacted catchments and reference catchments using two-tailed t-tests. Sites in AMD catchments that were upstream of AMD sources were included as reference sites.

	AMD	Reference	T-Stat	Degrees of Freedom	p-value	Significantly different?
Sulphate (mgL^{-1})	80.25 (± 24.06)	0.19 (± 0.10)	2.9	28	0.007	Yes
pH	Range 2.8-7.08	Range 5.2-7.8	-3.3	28	0.003	Yes
Conductivity (μScm^{-1})	289 (± 112)	88 (± 11)	1.56	28	0.13	No
Width (m)	5.0 (± 1.3)	8.0 (± 1.5)	-1.5	28	0.14	No
Depth (m)	0.4 (± 0.04)	0.6 (± 0.08)	-2.81	28	0.009	Yes
Channel stability score	68.8 (± 3.5)	67.0 (± 6.4)	0.27	28	0.79	No
Temperature (°C)	14 (± 1.2)	13 (± 1.0)	0.77	28	0.45	No
Dissolved Oxygen (mgL^{-1})	8.0 (± 0.5)	11.6 (± 0.9)	-3.6	28	0.001	Yes
Stream Order	2.7 (± 0.2)	3.1 (± 0.3)	-1.2	28	0.24	No

The Waimangaroa River, the longest AMD impacted system, had the most non-impacted tributaries entering it after AMD contamination (Table 2.4). Whereas, Miller Creek had no tributaries to dilute the effects of AMD, as all tributaries in the Miller Creek system were affected by mining activities. Charming Creek had the most diluting tributaries per kilometer of impacted stream-length, and therefore had approximately the highest dilution rate.

Table 2.4 The number of non-AMD impacted tributaries providing dilution to AMD impacted streams

AMD impacted stream	Distance of downstream site from source AMD (km)	Number of non-impacted tributaries
Waimangaroa River	8.5	16
Charming Creek	3.5	8
Miller Creek	3.5	0
Darcy Stream	3.5	5

Biological

Periphyton

AMD catchments had more organic biofilm on the substrate surface than reference catchments (Figure 2.7). Sites from AMD catchments that were upstream of AMD impacts were included in the mean for reference catchments, as these were effectively not impacted by AMD.

Macroinvertebrates

Elevated sulphate levels, and low pH are both common in AMD impacted streams. These parameters are correlated with macroinvertebrate data. Figure 2.8 shows that benthic invertebrate density is negatively correlated with sulphate concentration ($r = -0.391$), and strongly positively correlated with pH ($r = 0.626$). Sulphate and pH are also associated with the diversity of benthic invertebrates. Figure 2.9 shows that sulphate is negatively correlated with number of total taxa ($r = -0.422$), and pH is positively correlated with number of total taxa ($r = 0.658$).

The "L" shaped scatter in Figure 2.8, showing the correlation of sulphates and macroinvertebrate density suggests that streams with high densities of macroinvertebrates do not have sulphate, and streams with elevated levels of sulphate have few if any macroinvertebrates.

Mine impacted streams generally had communities that were lower in diversity and density, with fewer individuals per m^2 than reference streams. However, both reference and AMD catchments had modest numbers of taxa and densities, in comparison to other New Zealand rivers (Scarsbrook et al,. 2000).

Chasm Creek had low diversity, between 5 and 15 taxa at each site. Miller Creek also had low diversity, generally 5-10 taxa were present, except for the third site where less than 5 taxa were collected (Figure 2.10).

Tidal Creek and Charming Creek (upstream of AMD) had high numbers of taxa, generally more than 20 taxa. Tidal Creek maintains its diversity longitudinally, whereas in Charming Creek total number of taxa decreases considerably after it is impacted by AMD. However, the most downstream site in Charming Creek (furthermost from AMD), 15 taxa were recorded (Figure 2.10).

The Little Wanganui River showed a different pattern to the smaller reference streams. It had a higher diversity at the upstream sites (20-30 taxa), which declined downstream as the size of the river increased. The most downstream site had less than 10 taxa, possibly reflecting the changing morphology of the river as it approaches the sea. The Waimangaroa River had 10-15 taxa at its headwater sites, while the third site, which is in the closest proximity to the sources of AMD, has the lowest diversity (less than 10 taxa). The lowest site has a similar diversity to the most upstream site (Figure 2.10).

The three 'pollution sensitive' orders, Ephemeroptera, Plecoptera and Trichoptera (EPT) (Lenat, 1988), were the dominant taxa in the reference streams, in the Little Wanganui River EPT comprised of more 75% of the fauna, with the exception of the lowest site, where EPT comprised less than 25%. In Tidal Creek the benthic communities consisted of 70-100% EPT, a finding consistent along the length of the stream that was surveyed (Figure 2.11). Although low in density and diversity, the macroinvertebrates collected from Chasm Creek were predominantly EPT, with all sites having 80-100% EPT.

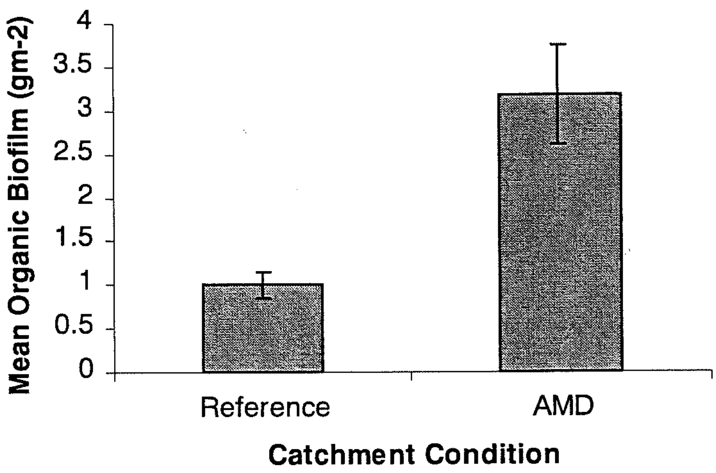


Figure 2.7 **Mean ash-free dry mass (+/-SE) of organic biofilm on the substrate for the AMD impacted and reference catchments. Sites in AMD catchments that were upstream of AMD sources were included in the mean for Reference sites, rather than AMD.**

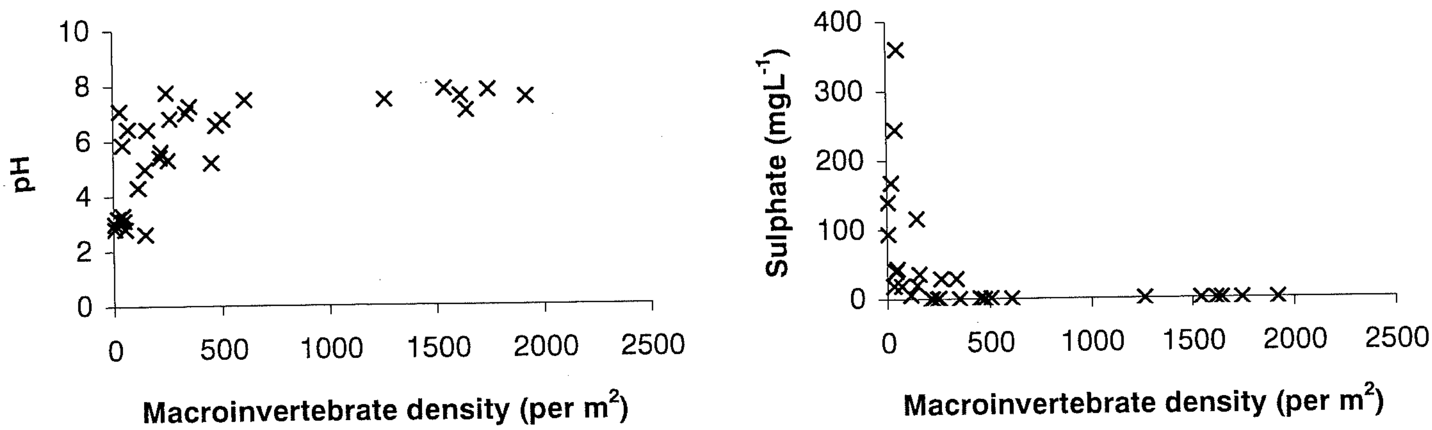


Figure 2.8 **Correlation of pH and sulphate with macroinvertebrate density.**

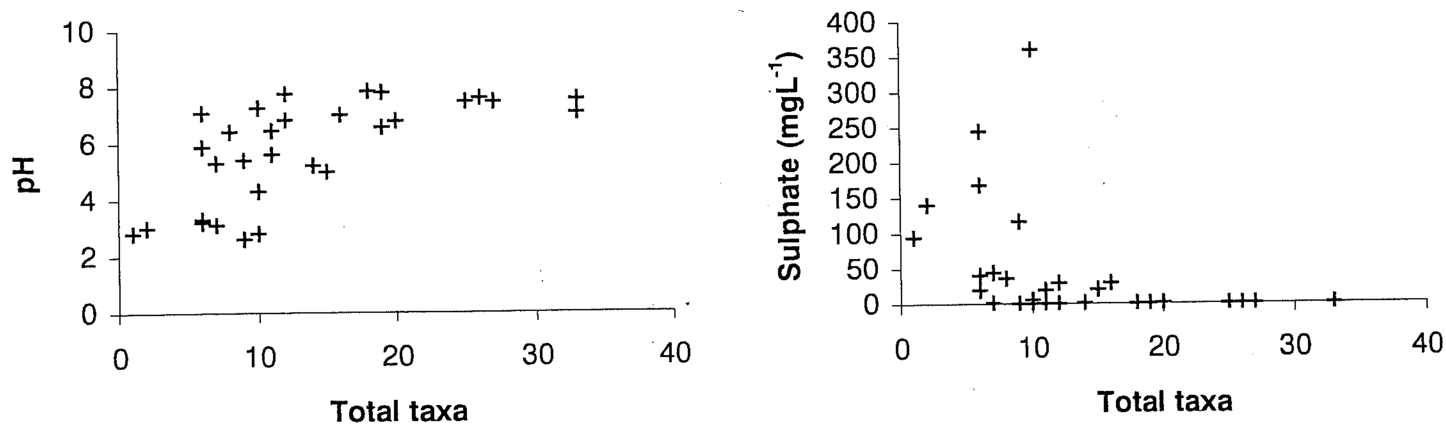


Figure 2.9 **Correlation of pH and sulphate with the number of total taxa foumd.**

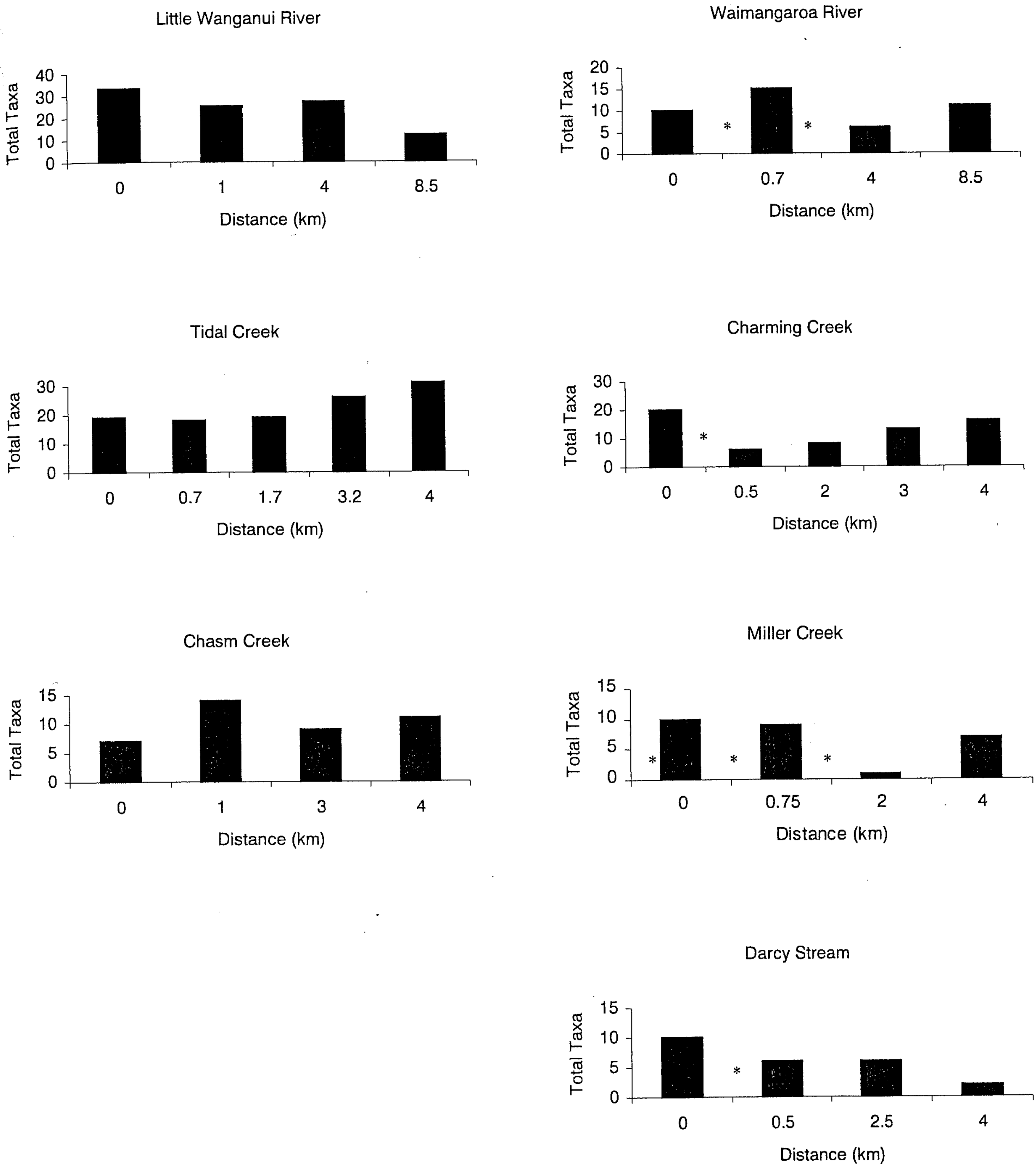


Figure 2.10 **Total taxa from three Surber samples and 1 kicknet sample of all habitat types at each longitudinal site. Each site is represented by its downstream distance from the first site. In the impacted streams, the point of AMD impact is marked "*".**

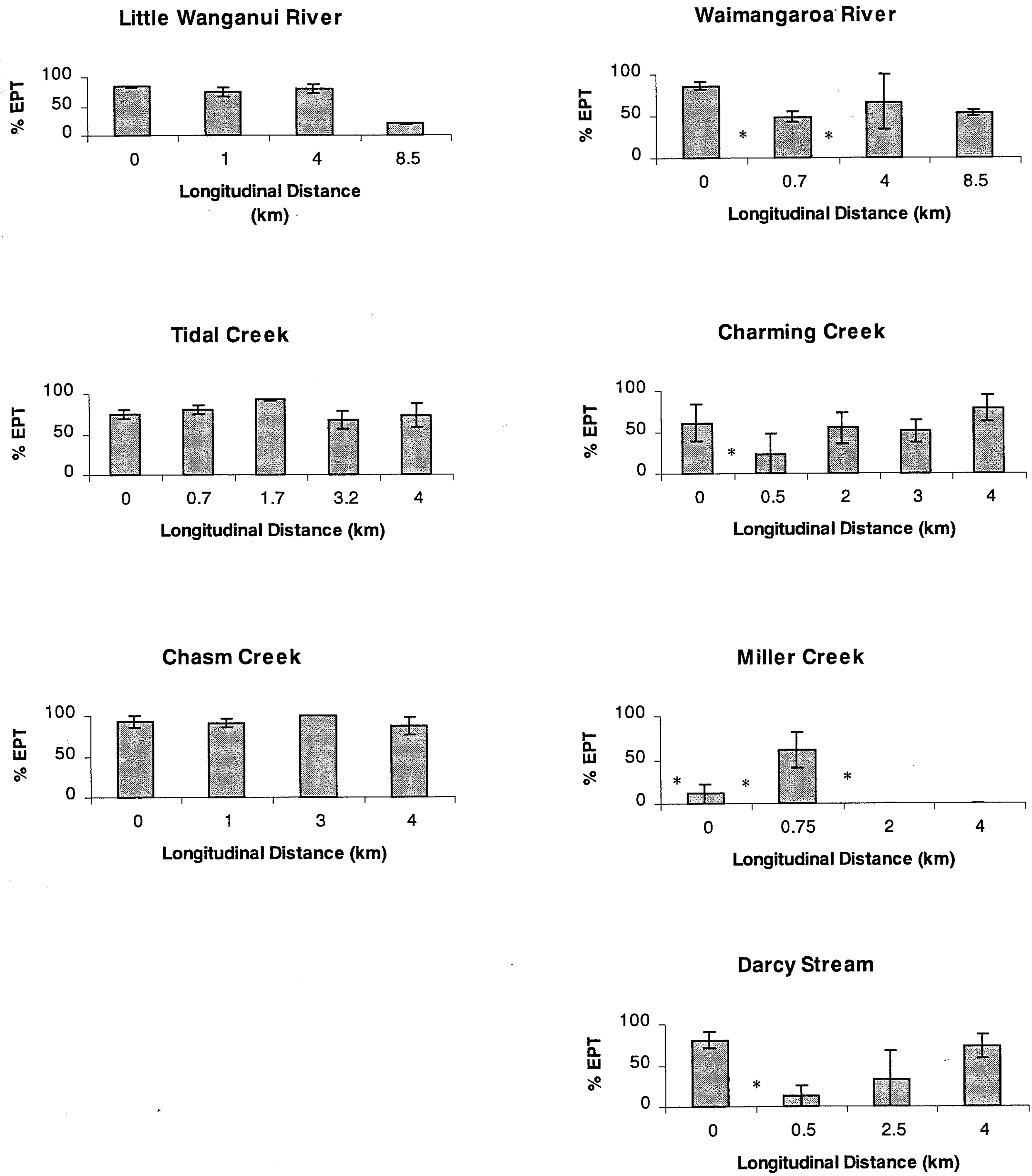


Figure 2.11 The percentage of total fauna/m2 (+/- SE) that is comprised of Ephemeroptera, Plecoptera and Trichoptera (EPT). Each site is represented by its downstream distance from the first site. In the impacted streams, the point of AMD impact is marked

In contrast, mine impacted catchments had a much lower proportion of their benthic communities as of EPT taxa. The headwater site of the Waimangaroa River, which was not AMD impacted, 80% of the community were EPT, but the downstream sites had <50%. Similarly, Charming Creek had less EPT downstream of AMD effluent entering the stream, although the proportion increased, and was similar in the most downstream site to that of the first site (pre-AMD) (Figure 2.11).

Miller Creek generally had a very low percentage of EPT individuals (<20%). The second site had 50% EPT, but as density was extremely low in this stream, this is likely to be greatly skewed by one or two individuals.

Darcy Stream had greater than 75% EPT above the mine input, indicating that these taxa dominated the community. The two subsequent sites downstream of where mine discharge entered the stream had greatly reduced %EPT (25-50%), although the most downstream site had 70%, almost the same level as the first site (pre-AMD) (Figure 2.11).

Overall, AMD impacted streams have macroinvertebrate communities that are reduced in density, and diversity and are generally dominated by taxa other than Ephemeroptera, Plecoptera and Trichoptera. Furthermore, AMD streams tend to have increased abundance of organic biofilm on the surface of the substrate, which reflects the lower densities of macroinvertebrates (Figure 2.12).

The Little Wanganui River had a comparatively high density of macroinvertebrates, although density was lower at the second site, and at the most downstream site where the river was large (Figure 2.12). The Waimangaroa River had lower densities than the Little Wanganui River. The most upstream site of the Waimangaroa River, which was not receiving mine contaminated water, had higher densities of benthic invertebrates than the three sites downstream of AMD inputs. These three sites had very low densities (<50 individuals/m^2) (Figure 2.12).

Tidal Creek generally had very high densities of macroinvertebrates (Figure 2.12). The most upstream site, a small 1st order tributary, had lower densities, but the four larger downstream sites had densities >1000 individuals per m^2. In contrast, Charming Creek had relatively high densities of invertebrates at the non-impacted site upstream of the AMD source, but downstream densities dropped markedly. However, as downstream longitudinal distance from the point of AMD impact increase, invertebrate densities also increase (Figure 2.12).

Chasm Creek had higher densities the upper sites, and density increased in the two lower sites. Abundances in this stream were very variable, but were always greater than 500 individuals per m^2. Although a similar sized stream, Miller Creek had much lower densities of macroinvertebrates than Chasm Creek. There was little variation in density between the longitudinal sites of Miller Creek (Figure 2.12).

Darcy Stream had a higher density of macroinvertebrates in the two most AMD impacted sites. Sites 2 and 3, immediately downstream of the source of AMD had significantly more individuals than Sites 1 and 4, an upstream non-impacted site, and the furthermost site from the source of AMD respectively (Figure 2.12). However, this contrasts with total taxa. Darcy stream had the most taxa upstream of AMD (Site 1). Downstream of AMD, the number of taxa decreased markedly, and remained low along the length of the stream (Figure 2.10).

A comparison of the relative abundance of major macroinvertebrate groups suggests that communities in reference catchments, and sites upstream of AMD impacts, were dominated by Ephemeroptera, particularly in headwater sites (Figure 2.13). Trichoptera were also relatively abundant in reference streams. In AMD impacted streams communities (sites downstream of AMD sources) Diptera generally had higher abundance in the community relative to other taxonomic groups in comparison to reference sites. However, in the Waimangaroa River, Charming Creek and Darcy Stream the relative abundance of Diptera decreases at the sites further downstream from the source of AMD. Instead, Trichoptera and Ephemeroptera become more dominant in the community (Figure 2.13).

The Berger Parker index of taxa dominance (Berger & Parker, 1970) indicated that reference streams were dominated by single taxa, and that this was relatively uniform between sites on a stream (Figure 2.14). In contrast, the AMD impacted sites were less uniform. Some streams were dominated by single taxa (eg. Miller Creek), but there was less dominance at others, such as Darcy Stream.

Taxon richness as described by Margalef's index indicates that the reference sites generally had higher and more uniform taxon richness (Figure 2.14). In comparison, sites in the AMD impacted catchments were more variable, and impacted sites had lower richness. The Waimangaroa River, Charming Creek and Darcy Stream showed marked reduction in richness after AMD impact, but this increased with longitudinal distance from the source of AMD. Miller Creek had very low richness at all sites.

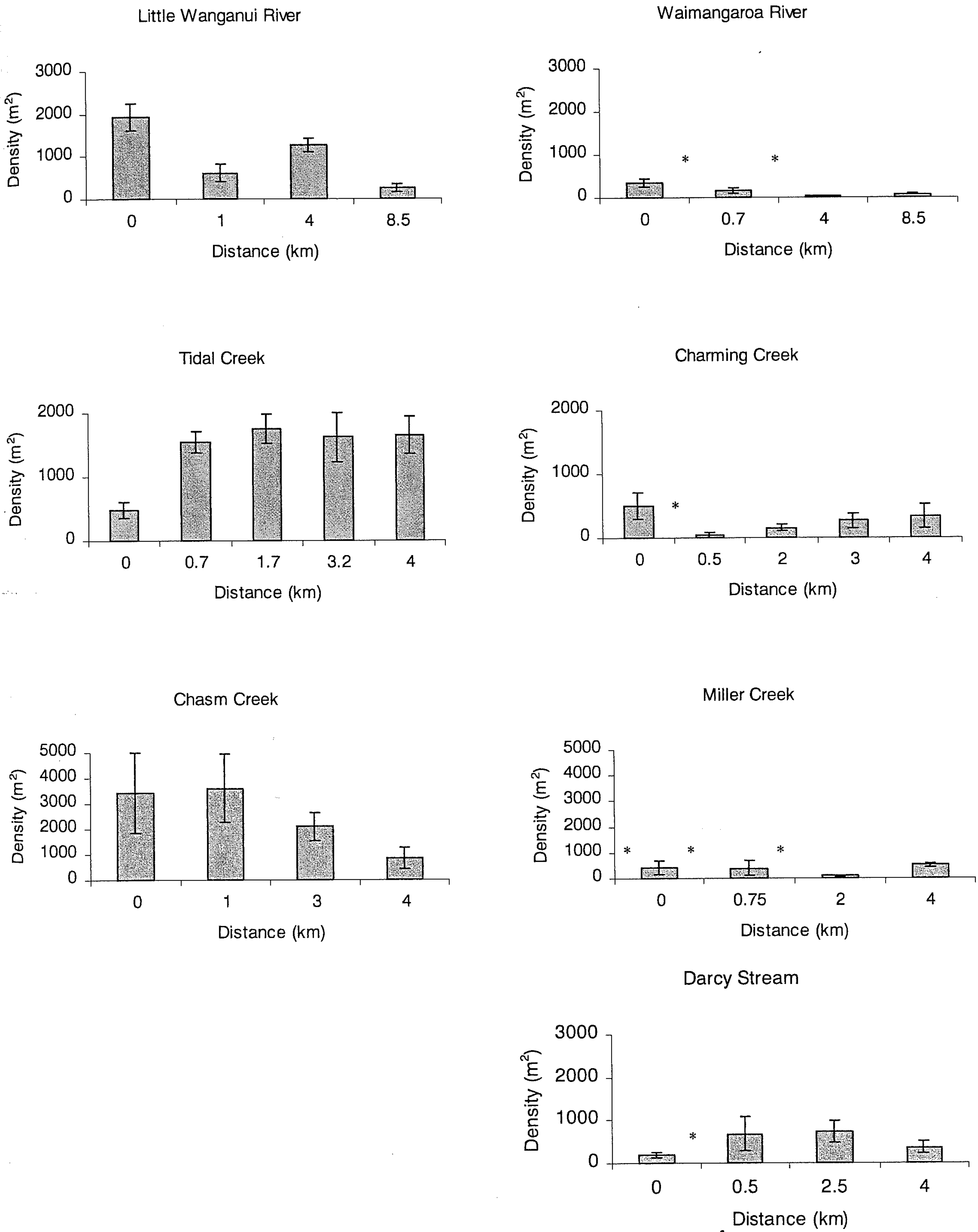


Figure 2.12 **Mean density (±SE) of macroinvertebrates (individuals per m^2) at longitudinal sites of surveyed catchments (NB x axis is not to scale). Each site is represented by its downstream distance from the first site. In the impacted streams, the point of AMD impact is marked "*".**

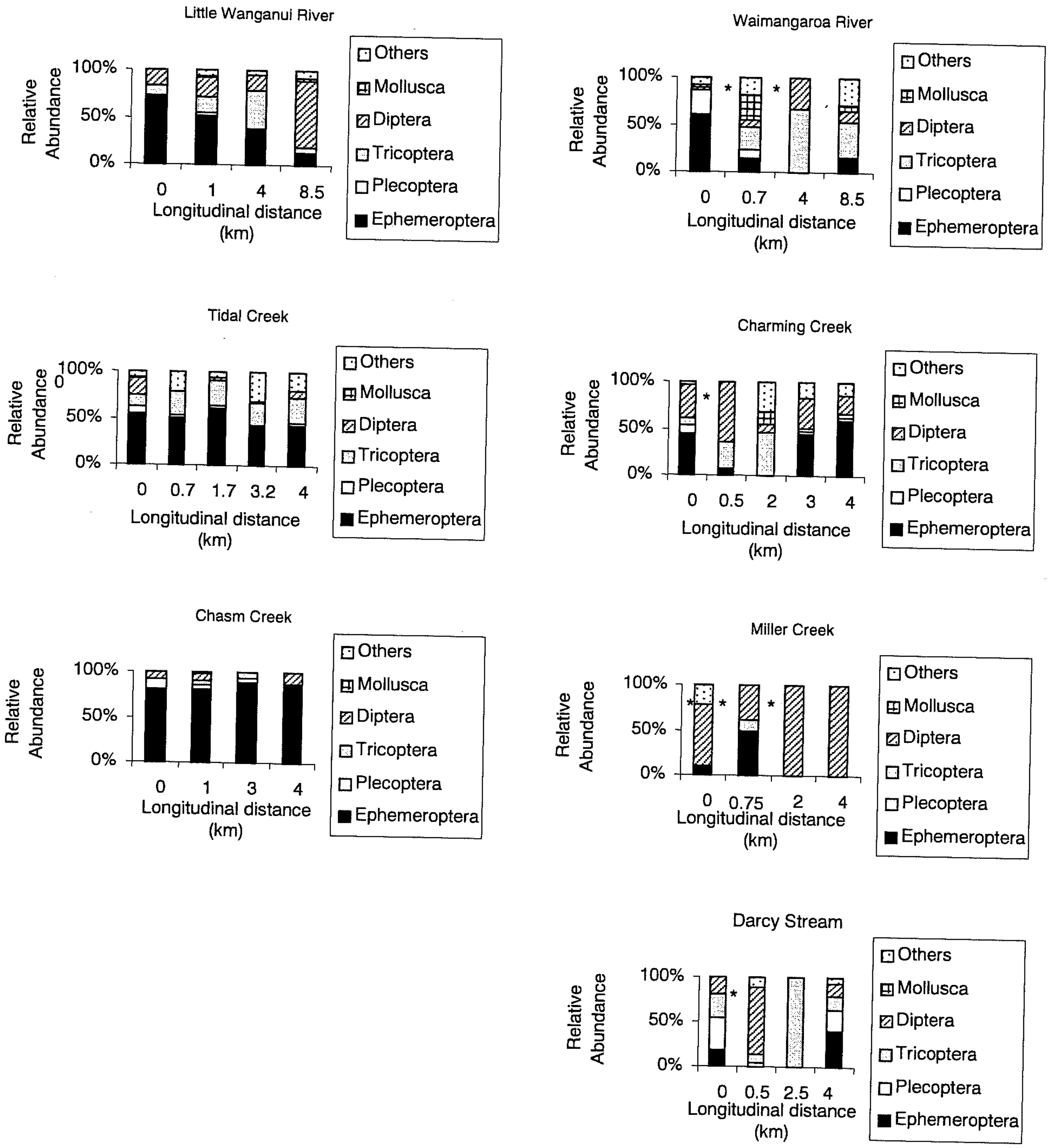


Figure 2.13 **Relative abundance of major taxonomic groups. * Indicates the point of AMD entry into the stream.**

Deleatidium mayflies were frequently the most abundant taxa, both in reference catchments and AMD impacted catchments (Table 2.5). The majority of *Deleatidium* abundance in AMD catchments was found in either sites upstream of AMD sources, or at the downstream-most sites.

Diptera taxa were frequently abundant taxa at AMD impacted sites. At Miller Creek and Darcy Stream Chironomidae taxa were numerically dominant, particularly Orthodcladiinae. Chironomids were numerically significant taxa in the Waimangaroa River and Charming Creek.

Oxyethira, a pollution tolerant caddisfly, was also frequently in the five most abundant taxa in AMD impacted catchments, as was *Zelandobius*, a stonefly that is tolerant of low pH.

In the reference catchments, mayflies (*Deleatidium, Nesameletus*), caddisflies (*Helicopsyche, Aoteapsyche, Olinga*) and Elmidae beetles were frequently amoungst the numerically dominant taxa (Table 2.5).

Table 2.5 **Five most abundant taxa in each catchment.**

Catchment	Dominant taxa				
Little Wanganui River	*Deleatidium* (Ephemeroptera)	*Aoteapsyche* (Trichoptera)	*Helicopsyche* (Trichoptera)	Elmidae (Coleoptera)	*Aphrophila* (Diptera)
Waimangaroa River	*Deleatidium* (Ephemeroptera)	*Zelandobius* (Plecoptera)	*Oxyethira* (Trichoptera)	Elmidae (Coleoptera)	*Cricotopus* (Diptera)
Tidal Creek	*Deleatidium* (Ephemeroptera)	*Olinga* (Trichoptera)	*Nesameletus* (Ephemeroptera)	Elmidae (Coleoptera)	*Helicopsyche* (Trichoptera)
Charming Creek	*Deleatidium* (Ephemeroptera)	*Oxyethira* (Trichoptera)	*Nesameletus* (Ephemeroptera)	Orthocladiinae (Diptera)	Elmidae (Coleoptera)
Chasm Creek	*Deleatidium* (Ephemeroptera)	*Nesameletus* (Ephemeroptera)	Elmidae (Coleoptera)	*Zelandobius* (Plecoptera)	*Cricotopus* (Diptera)
Miller Creek	Orthoclad Sp A (Diptera)	Orthoclad Sp VIII (Diptera)	*Polypedilum* (Diptera)	*Deleatidium* (Ephemeroptera)	*Maoridiamesa* (Diptera)
Darcy Stream	*Eukiefferiella* (Diptera)	*Oxyethira* (Trichoptera)	*Zelandobius* (Plecoptera)	*Deleatidium* *Zephlebia* (Ephemeroptera) Tanypodinae (Diptera)	

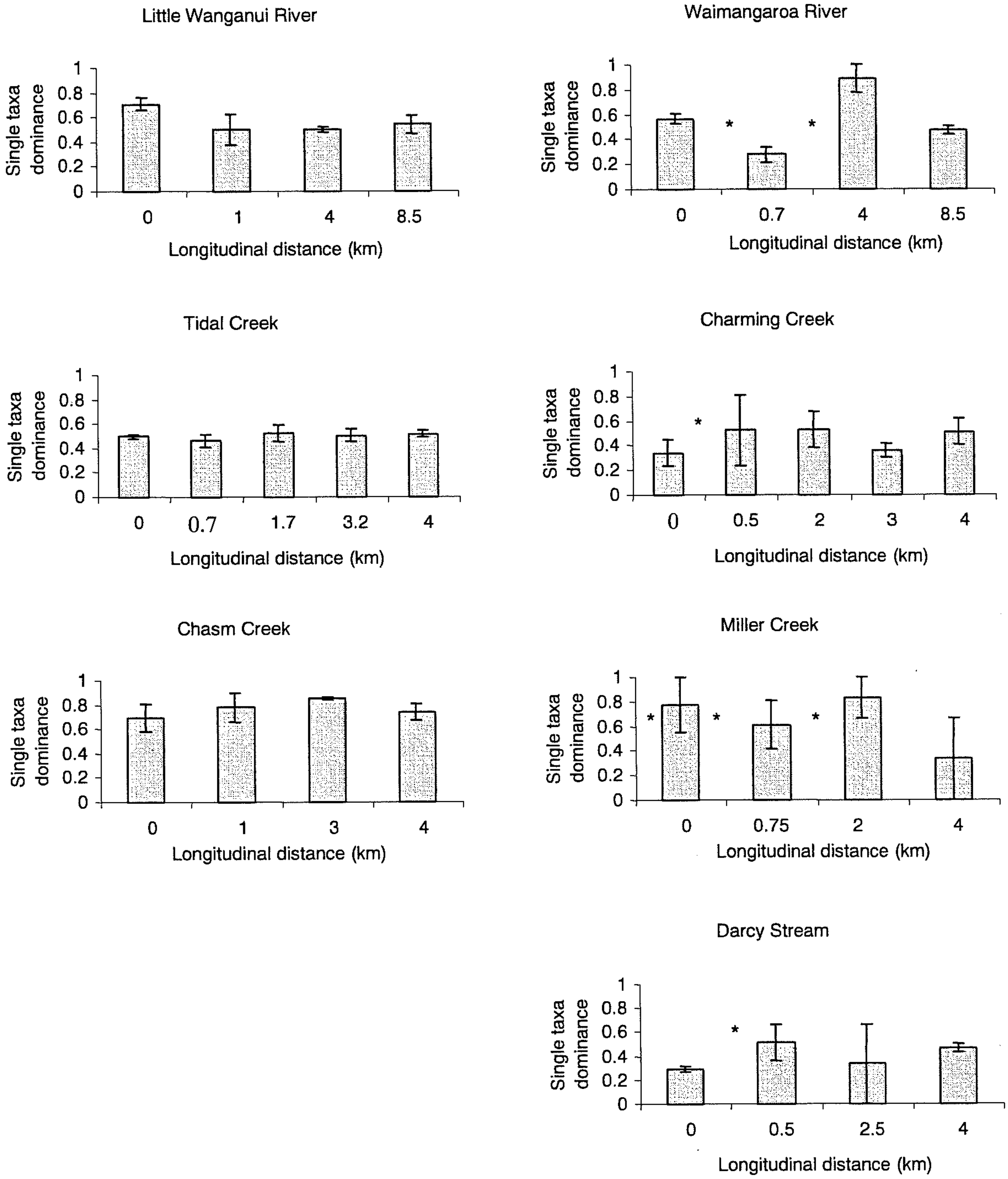


Figure 2.14 **Mean dominance (± SE) of single taxa at each site, quantified with the Berger Parker index. * Indicates the point of AMD entry into stream.**

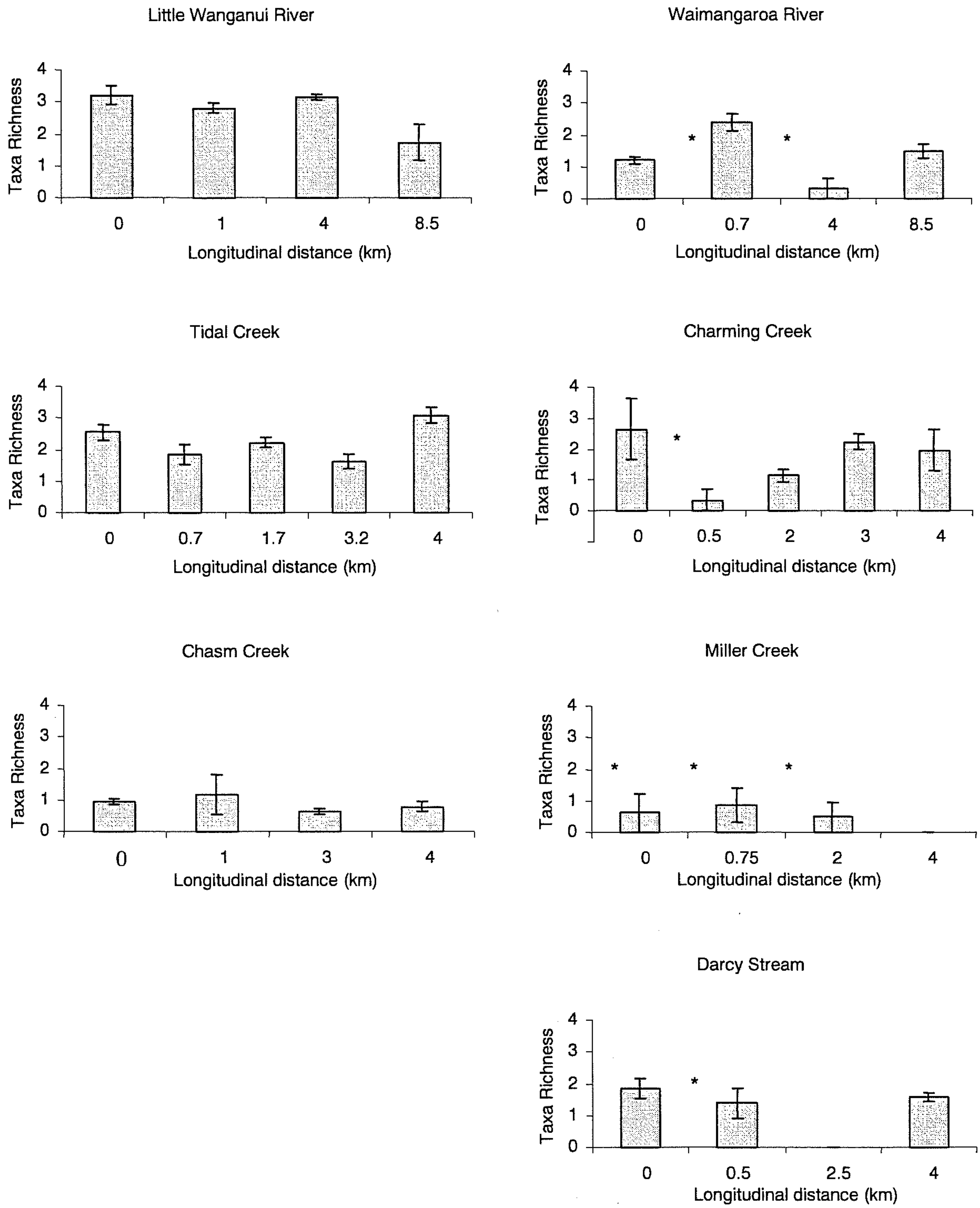


Figure 2.15 Mean Margalef's index scores (±SE) for taxa richness. * Indicates AMD entry into stream.

A multi-variate cluster analysis of taxon presence/absence for all sites generally grouped most of the AMD impacted sites together (Figure 2.16). The third site of Darcy Stream, and the first, third and fourth sites of Miller Creek were initially separated from the other sites. The remaining sites from Darcy Stream and Miller Creek were grouped with the second site of Charming Creek (the first AMD impacted site), and the two lower Waimangaroa River sites.

The first Charming Creek site (pre-AMD) was grouped with the lowest site, suggesting these two sites are more similar to each other than any other Charming Creek sites. Site 4 was the next closest.

The reference sites were relatively scattered and did not appear to be clustered with any distinct pattern, except for the highest Tidal Creek site, and the lowest Little Wanganui site.

A TWINSPAN analysis of the density of macroinvertebrate taxa (Figure 2.17) indicated that most of the AMD impacted sites were similar. After a fourth iteration, the remaining contaminated sites were divided into two groups. These were significant, as they reflected the proximity of the site to the point source of AMD contamination. In the negative group, for which the preferential species were the pollution-tolerant caddisfly *Oxyethira* sp, and *Collembola*, consisted of the third site of the Waimangaroa River (receiving more AMD effluent than the other Waimangaroa sites), the second site of Charming Creek (the nearest site to Wearne Creek, which discharges AMD into the Charming Creek mainstem), the upper two sites on Miller Creek (both in close proximity to several old mine workings), and the second Darcy Stream site (immediately downstream of AMD discharge).

The positive group, characterised by the stonefly genus *Zelandobius* and the mayfly *Nesameletus*, grouped the highest (non-contaminated) and lowest Waimangaroa River and Darcy Stream sites together. The first two Chasm Creek sites were also in this group, as were the first (non-contaminated) Charming Creek site, and the fourth.

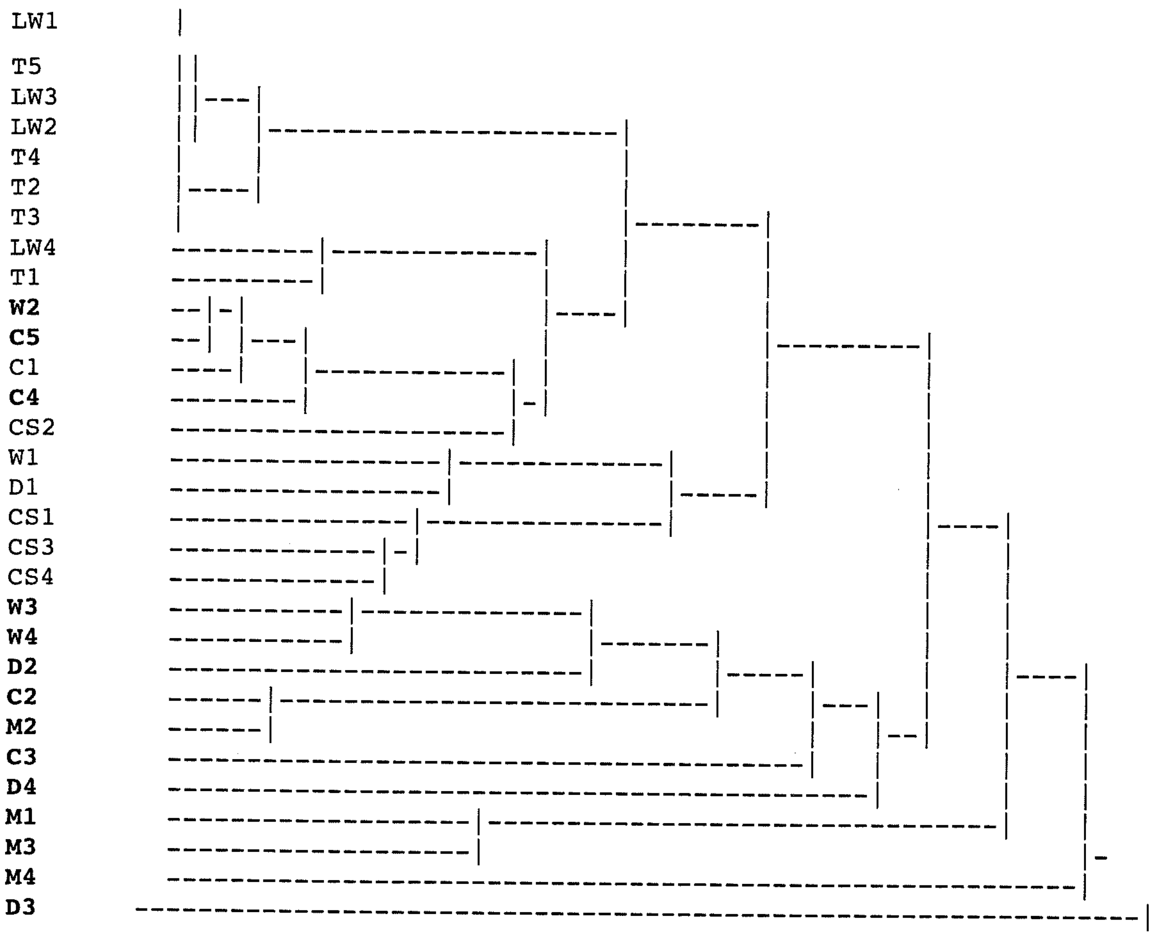


Figure 2.16 **Cluster analysis of density data. AMD impacted sites are in bold. Site codes refer to the first letter of the Stream name (C = Charming Creek, Cs = Chasm Creek), and the number of the longitudinal site, with the most upstream site being Site 1.**

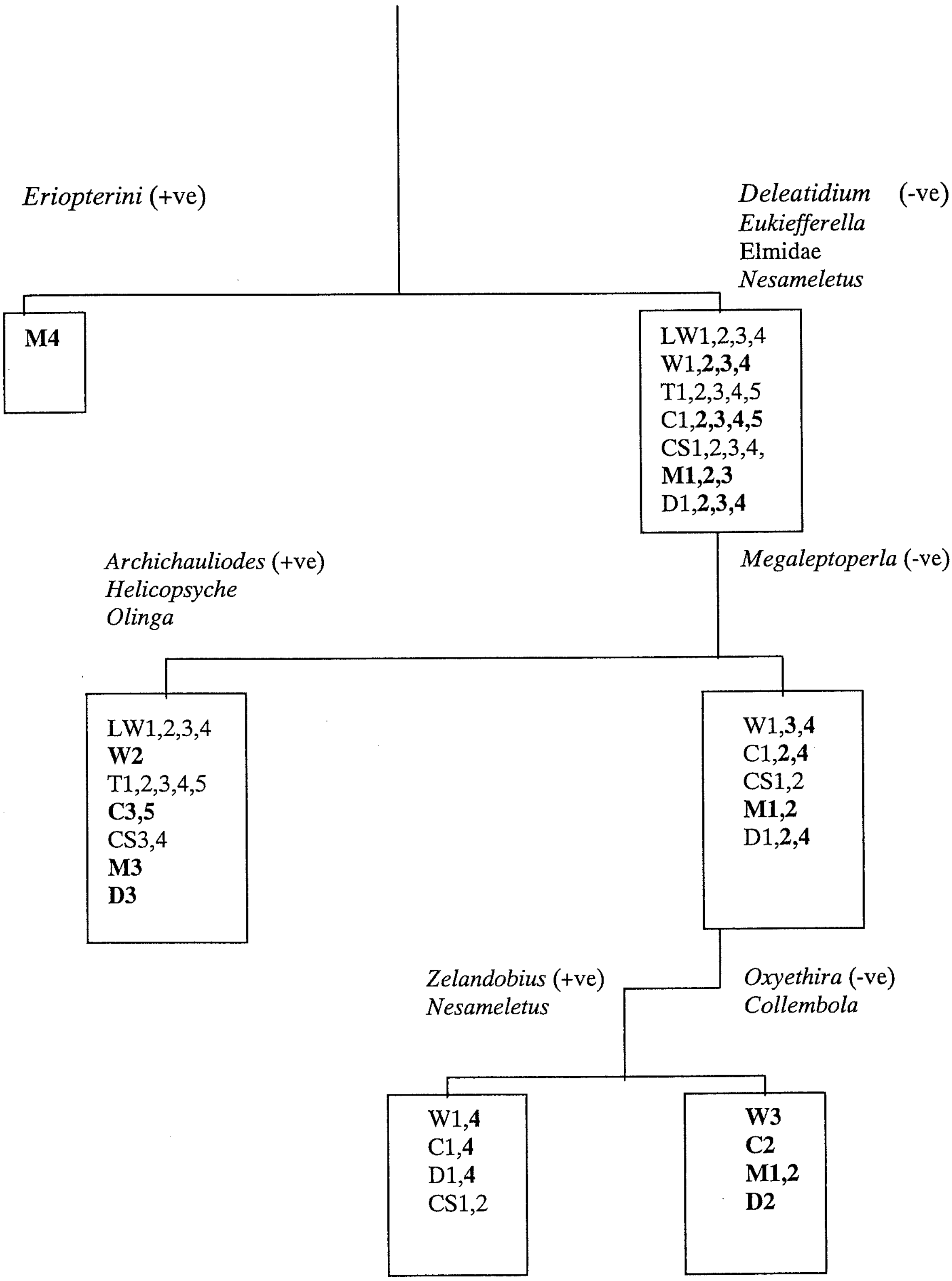


Figure 2.16 **Longitudinal survey site separation by TWINSPAN analysis of taxa abundance data. Indicator taxa are shown at each bifurcation. Site codes refer to the first letter of the Stream name (C = Charming Creek, Cs = Chasm Creek), and the number of the longitudinal site, with the most upstream site being Site 1. AMD impacted sites are shown in bold.**

DISCUSSION

The climatic, geological and physico-chemical conditions in catchments in the Buller Region create in-stream conditions that strongly influence the benthic invertebrate communities, even in catchments that are not greatly modified or affected by anthropogenic influence.

The Stockton/Denniston plateau area, where most of the streams in this study originate, has an annual rainfall of 2000-8000mm (West Coast Regional Council, 2003). This high rainfall leads to streams being prone to frequent flood disturbance. When combined with a topography that is typically very steep, these conditions result in streams that are highly disturbed environments, lacking substrate stability, and likely to be lacking zones capable of retaining allochthonous organic matter. These factors all contribute to reducing macroinvertebrate abundance and diversity.

Water velocity, temperature, substrate stability, and stream order were generally similar in both AMD and reference catchments, yet water chemistry and macroinvertebrate community composition were markedly different. This is important, as physico-chemical factors can profoundly affect the benthic invertebrate communities (Anthony, 1999; Collier et al., 1998). The similarity observed in this study gives some assurance that differences in community structure in the two types of catchment are likely to be due to the influence of acid mine drainage.

Low pH and elevated sulphate concentrations are both common indicators of contamination by AMD effluent in streams on the West Coast. This is a result of oxidisation of pyrite in the Brunner Coal Measures, releasing hydrogen and sulphate ions into the water (Collier et al., 1990; Edguardo, 1997). It is therefore to be expected that these are lower and higher respectively in AMD impacted waterways.

Dissolved oxygen is significantly lower in AMD impacted water. This is unusual, as there is no difference in temperature or water velocity between the two catchment types, and these are parameters that are usually associated with dissolved oxygen levels. Altitude also can affect dissolved oxygen, but selection of study sites ensured that altitudes were comparable between reference and AMD impacted catchments. The deficit in dissolved oxygen in AMD impacted streams is possibly explained by the addition of deoxygenated water entering streams from mine workings, or there may be a chemical process which reduces the availability of dissolved oxygen in the water column associate with acid mine effluent. Autotrophic production (algae and macrophytes) also affects dissolved oxygen levels. However, a significantly greater mass of organic matter was removed from substrate in AMD impacted streams, so this does not explain the lower dissolved oxygen in AMD streams.

The greater biomass of organic matter in AMD impacted streams may be due to a lower density of macroinvertebrates, and therefore a release of grazing pressure. However, the method used for quantifying organic matter did not allow for a distinction between autotrophic algal growth and heterotrophic organisms such as bacteria and fungi. Therefore, the difference in organic matter may not be a function of decreased grazing pressure, but instead a reflection of a change in species composition of biofilms. The iron-rich environment produced by AMD can encourage blooms of ferrophilic bacteria. It is not clear whether it is because the iron oxide displaces the competitive periphyton to allow these blooms, or if the iron oxide facilitates the bacterial bloom, subsequently displacing periphyton (Wellnitz & Sheldon, 1995). However, regardless of the mechanism, it is apparent that acid mine drainage affects the entire ecosystem of a stream, including primary production. This is subsequently likely to be having bottom-up effects on the entire food web, directly limiting grazing invertebrates, and indirectly restricting their predators.

The correlation of pH, and sulphate, with macroinvertebrate density and diversity suggests a strong association between these chemical parameters and benthic communities. Although pH and sulphate may not be directly affecting invertebrates, particularly in AMD catchments where pH is not markedly reduced (eg the Waimangaroa River), either these factors, or associated components of AMD impacted waterways (dissolved metals, altered substrate condition) are affecting the invertebrate community. Usually this results in causing a reduction in diversity and abundance of invertebrate taxa.

The diversity of macroinvertebrates in all the waterways surveyed was low, even in non-impacted catchments. Seldom were more than 30 different taxa collected at a site, even in catchments like those of Tidal Creek and the Little Wanganui River that are almost entirely without anthropogenic influence. This is a common phenomenon in West Coast streams. Relatively pristine streams have been found to have 10-30 taxa despite a lack of obvious human-influenced limitations (Harbrow 2001; Collier & Winterbourn, 1987).

For example, two reference sites, the most upstream site of Tidal Creek, and the most downstream site of the Little Wanganui River were singled-out by the cluster analysis. Both these sites had low densities of individuals, and diversity was constrained as a result. Both sites had different morphologies to the other sites on their respective streams. The top Tidal Creek site was much smaller than the other

Tidal Creek sites, being a first order tributary, and the lowest little Wanganui Site was much larger than the higher sites, and had reduced habitat heterogeneity.

A combination of harsh climatic conditions, a high frequency of disturbance events, profusion of humic matter and organic acids and unique geo-chemical conditions results in limitations on the diversity of macroinvertebrates in West Coast streams (Winterbourn & Collier, 1987).

The high rainfall of the area may be facilitating recovery and recolonisation of AMD impacted streams by macroinvertebrates, as high rainfall may dilute AMD water sufficiently to allow sensitive taxa temporal respite periods, during which previously inaccessible habitats can be invaded. Experimental work has suggested that intermittent influxes of AMD into a stream system are far less damaging to the ecosystem that continual AMD inputs (Soucek et al, 2000). Even when the chemical conditions during the periods of AMD flux are equally detrimental to those of the continuous impact system, the macroinvertebrate community proved to be relatively resilient (Soucek et al., 2000). This may be due to the fact that many of the toxic components of AMD affect macroinvertebrates at particular stages of their life cycle. For example, if AMD affects reproduction, this is unlikely to affect a population if it can breed in between periods of flux, or when rainwater provides sufficient dilution. Likewise, the physical effects of iron oxide precipitate can clog the gills of organisms like mayflies, but periodic moulting can clear these. Continual pressure of AMD would lead to this being a major limiting factor, but periods of temporal refuge may abate this.

In the Waimangaroa River and Miller Creek, the impact of AMD markedly reduces the density of macroinvertebrates, which remains very low and shows little improvement, even though the lower sites of the Waimangaroa River are more than six kilometres downstream of the source of AMD. However, the number of taxa in the Waimangaroa does show some longitudinal recovery, in that it resumes some of the characteristics of sites upstream of the AMD impact; and diversity is the same in the lowest site as it is in the site upstream of the AMD. Charming Creek also show signs of longitudinal recovery. Charming Creek has a marked reduction in both diversity of taxa and density upon the entry of AMD to the ecosystem. The community appears to recover, and 3.5 km from the source of AMD the macroinvertebrate community in Charming Creek is very similar to what it is upstream of the source of AMD impact. This further supported by the EPT data. Ephemeroptera, Plecoptera and Trichoptera are considered good indicator taxa of water quality A combination of harsh climatic conditions, a high frequency of disturbance events, profusion of humic matter and organic acids and unique geo-chemical conditions results in limitations on the diversity of macroinvertebrates in West Coast streams. In Charming Creek the proportion of EPT in the community decreases with the addition of AMD water, but gradually increases again downstream. This indicates that the community composition is gradually reverting to a similar structure to that of areas upstream of AMD. A prevalence of sensitive taxa like mayflies, stoneflies and caddisflies suggests that the community is relatively healthy and the stream is only mildly impacted (Lenat, 1988).

The response of the macroinvertebrate community in Darcy Stream is more complicated. Initially, the density of macroinvertebrates increases individuals after AMD impact. This may be partially explained by the release of a few AMD tolerant taxa (for example, some Chironomidae taxa), from competition and predation, allowing populations to expand (Allard & Moreau, 1986). The diversity of the community in Darcy Stream reflects this. It declines significantly after AMD enters the stream, and remains low, despite the increase in density. Chironomid taxa (eg *Eukiefferiella*), particularly increased. The EPT component of the Darcy Stream community is significant upstream of the source of AMD (>70%), but consists of less than 25% downstream of the initial point AMD impact. However, with increasing longitudinal distance the community composition recovers, and in the lower catchment EPT individuals again are a significant component of the community (>70%), even though total densities of macroinvertebrates are comparatively low.

The AMD impacted sites all have similar benthic communities, and were grouped accordingly in the cluster analysis. This analysis did not show any clear patterns of longitudinal change, with the exception of Charming Creek. The cluster shows that the two lowest sites were more similar to the site upstream of AMD impact, suggesting that the community is returning to a similar structure with longitudinal distance. Charming Creek has several small tributaries downstream of the source of AMD which are likely to be diluting the contaminated water sufficiently to allow this recovery. Charming Creek has the most tributaries per kilometre of impacted stream length. Accordingly, the water chemistry of Charming Creek shows improvements with longitudinal distance. pH was circum-neutral at the lowest site, as it is at the site upstream of the AMD source, whereas it drops below 6 immediately downstream of the source of AMD. Similarly, the level of sulphate in Charming Creek decreased by 11 mgL^{-1}, from 40 mgL^{-1} immediately downstream of the mine effluent to 29 mgL^{-1}, less than four kilometres downstream. As sulphate is conserved in the system, it is likely that dilution, rather than any chemical process or precipitation, that has caused this decrease. It therefore it follows that other contaminants such as dissolved metals are similarly reduced in concentration, the net result being an improved chemical environment for macroinvertebrates. The physical channel remains very similar in Charming Creek for its entire length, and it is almost certainly this improvement in water quality that facilitates a recovery in the benthic macroinvertebrate community.

The presence of uncontaminated tributaries is doubly important, as not only do they facilitate a dilution of AMD water, they provide a source of colonists for impacted streams (Malmqvist & Hoffsten 2000). This is particularly important for several reasons. If longitudinal distance and dilution produce suitable habitat for macroinvertebrates, communities may still be limited by the lack of colonists. Heavily contaminated reaches upstream are very likely to be a chemical dispersal barrier for invertebrates that might otherwise have colonised from stream reaches upstream of AMD by drifting. Furthermore, if a rainfall event temporarily improves water quality, invertebrates can recolonise and exploit previously inhabitable areas and food sources quickly (Soucek et al., 2000).

The TWINSPAN analysis considered abundance as well as diversity, and it suggests that AMD impacted communities are recovering with increased longitudinal distance from the source of AMD pollution, with the exception of Miller Creek. Miller Creek is a very short catchment, and the area studied was less than four kilometres in length. The water chemistry of this stream indicates that it is severely impacted, and the invertebrate community reflected this. There were very few individual macroinvertebrates collected from Miller Creek, rendering any analysis unreliable, and the ability to detect any community changes poor. Due to extensive current and historic mining activity in the entire Miller Creek catchment there are a lack of uncontaminated tributaries to dilute AMD water and improve water quality.

The macroinvertebrate communities of Charming Creek, Darcy Stream and the Waimangaroa River all exhibit longitudinal recovery. The TWINSPAN analysis grouped the upstream, uncontaminated sites with the furthermost downstream sites, and separately grouped the sites immediately downstream of AMD impact. This return to a similar community structure also suggests that ecosystems also function in a similar manner to those upstream of AMD effluent.

The mean pH of all contaminated sites surveyed in this study was 4.9, which is comparatively high by the standards of many AMD impacted streams in the area (eg Rapid Creek, pH 2.8). Only Darcy Stream and Miller Creek had pH levels of less than 4.5. Therefore, macroinvertebrates in Charming Creek and the Waimangaroa

River, in heavily contaminated sites were not necessarily contending with more toxic pH levels than the natural background pH of the area. West Coast AMD impacted streams with pH levels between 4 - 5 tend to have more diverse communities than severely contaminated streams. Usually 9-13 taxa are present, as opposed to 2-6 taxa in streams with a pH below 3. Of these taxa in moderately contaminated sites, salient taxa include stoneflies such as *Spaniocercoides* and *Zelandobius* (Harbrow, 2001). This study also found stonefly taxa at moderately contaminated sites, such as the first contaminated site in the Waimangaroa River where there were an abundance of aquatic bryophytes.

The effects of AMD are evidently more complex than a simple case of a loss of diversity or abundance. However, the loss of even one or two species from a system can alter the dynamics of a community, and have a cascading effect. For example, the TWINSPAN analysis suggests that the lack of *Archichauliodes* is an important indicator associated with AMD impacted streams. *Archichauliodes* is a predator in its later instars, and the loss of such a species may allow an increase in prey species like chironomids. In turn, this can put increased competitive pressure on other taxa that belong to the same functional feeding group that may already be suffering from the sub-lethal affects of AMD. Harbrow (2001) found that there was a shift in the dominance of functional feeding groups. Where pH levels are less than 4.5, predators and algal grazers are frequently more prevalent, while collector-browsers become less dominant. Many invertebrates can become increasingly susceptible to predation following prolonged exposure to the products of acid mine drainage. The effect of contaminants, particularly heavy metals, can lead to altered behaviour, ultimately reducing the resilience of individuals to predation (Clements 1999). This may partially explain an increase in the dominance of predators. As prey become more easily captured, foraging by predation will improve in efficiency and facilitate a potential increase in predatory taxa. However, New Zealand benthic invertebrates are predominantly generalists in habitat and niche selection (Winterbourn et al., 1981). Macroinvertebrates frequently show capabilities for transferring foraging efforts to alternative trophic levels and feeding mechanisms when required. Furthermore, New Zealand benthic macroinvertebrates are also able to withstand a highly variable, and at times extreme, range of physico-chemical conditions like flooding, unstable substrata, and fluctuations in temperature and pH. Populations of benthic macroinvertebrates persist in this environment either through adaptation, tolerance or rapid decolonisation. This generalist habit enables many taxa to colonize AMD impacted streams that are unusually inhospitable. West Coast streams with pH levels less than 3.6 resulting from AMD contamination have been found to have Trichoptera species living in them (Winterbourn, 1998; this study). Two mayfly genera, *Deleatidium* and *Austroclima,* have been found living in streams with pH levels that are less than 4.5, conditions that would normally be expected to exclude sensitive taxa like Ephemeroptera and Trichoptera (Winterbourn, 1998).

Not relying on a single food source is an advantage in an AMD impacted stream. Leaves decompose very slowly, and are only retained in streams for short periods (Harbrow, 2001). Therefore, allochthonous inputs in such streams are likely to spasmodic and unreliable. Furthermore, the precipitation of iron hydroxide is associated with a decrease in the standing algal crop (Niyogi et al., 1999). Therefore, the ability of a macroinvertebrate to adjust its feeding according to availability is likely to be an advantage in a system where periphyton, coarse organic matter and fine organic matter can all be limited.

Therefore, AMD may be impacting stream communities partly with acute toxicity, but also by altering benthic communities with mechanisms such as top-down effects, and therefore affecting the corresponding community function.

However, the results of this study suggest that recovery from these community affects is possible. Charming Creek, Darcy Stream and the Waimangaroa River all show at least a partial return to the non-impacted stream community composition downstream of sources of acid mine drainage. Several factors are likely to be crucial to this recovery. Firstly, the proximity of healthy tributaries to a recovering reach provide a ready source of colonists in the event of improvements in water quality, however temporary. Tributaries also can provide a temporal refuge for invertebrates during periods of AMD flux. Furthermore, tributaries influence the abiotic environment of a river system (Rice et al., 2001), and in the case of AMD, provide dilution of effluent which improves water quality and reduces the toxicity of the contaminants.

The chance of encountering non-impacted tributaries is greatly improved by the length of the catchment. Therefore, longer waterways like Darcy Stream and the Waimangaroa River are expected to have a higher probability of showing recovery in the macroinvertebrate community. This partially explains why no recovery was apparent in Miller Creek. The Miller Creek catchment is relatively small, and is in close proximity to the sea. Therefore, any tributaries entering Miller Creek are equally contaminated and provide none of the advantageous dilution or refugia.

The initial level of contamination is also likely to an important factor in longitudinal recovery. Moderately contaminated streams can maintain a significant component of their invertebrate diversity (Harbrow, 2001). Therefore, recovery is more a case of an increase in abundance as water quality improves downstream, rather than replacing species that are completely absent.. This is a further reason as to why Miller Creek did not show longitudinal recovery. This stream had very low levels of pH, high conductivities and high concentrations of sulphates, suggesting it is heavily impacted and unlikely to recover without active remediation.

Although none of the AMD impacted catchments studied showed a complete recovery of benthic macroinvertebrate communities with longitudinal distance, this study suggests that at least partial recovery of diversity and sensitive taxa does occur given the appropriate initial conditions.

Chapter Three

The mechanisms of acid mine drainage toxicity

Dissolved and precipitated metals can occur naturally in freshwater environments, as a result of a variety of natural processes. These include the weathering of metal-bearing rock, geothermal and volcanic activity, as well as atmospheric inputs from precipitation, and even sea-spray (Smith & Williamson, 1986). Some metals, such as chromium, copper, iron, nickel and zinc are essential for life in trace amounts. However, all metals are potentially toxic to organisms when concentrations exceed tolerance thresholds (Smith & Williamson, 1986), although non-essential metals, such as arsenic, cadmium, mercury, and lead, can be withstood by organisms at low concentrations (Larcombe, 1986).

Mineral extraction activities can result in elevated ambient concentrations of metals in both the water and sediments of waterways. Streams that receive drainage from mine workings and mine tailings frequently show high concentrations of metals (Alarcon Leon & Anstiss, 2002; Beltman et al., 1999; Dills & Rogers, 1974). Acid mine effluent can contain high concentrations of both ferric and ferrous iron, mercury, arsenic, cadmium, copper, zinc, lead, aluminium and magnesium (Parsons, 1977; Stenson et al., 1993). For example, some mine-impacted streams in the Coromandel have been found to have cadmium concentrations as high as 9.3 mg m^{-3}, and zinc concentrations of 5950 mg m^{-3}. The US EPA (1980) considers the criteria for the protection of aquatic life to be concentrations of cadmium and zinc to be less than 1.5 and 180 mg m^{-3} respectively (Beaumont et al., 1987). High concentrations of metals also accumulate in the sediment. In the extensively mined Fly Creek watershed in Virginia, USA, iron concentrations in the sediment are as high as 10 000 mg kg^{-1}, and aluminium concentrations as high as 1500 g kg^{-1} (Cherry et al., 2001).

On the West Coast of the South Island, the Brunner Coal Measures are mined extensively. These coal measures have a high concentration of iron pyrite and other iron sulphide minerals (Harbrow, 2001). The process of mining exposes pyrite and causes oxidation, which releases metal ions and hydrogen irons into the ground water. These dissolved ions are then transported into streams and rivers in catchments where mining has occurred, both historically and recently. The waterways become contaminated as the pH is lowered by the addition of the hydrogen ions, and by the potentially toxic concentrations of dissolved and precipitated metals in the water and sediment (Fajtl et al., 2002). The compounding effects of other factors may also increase the impact of metal contamination. For example, increasing altitude increases the sensitivity of invertebrates to metal toxicity (Clements & Kiffney, 1995; Kiffney & Clements, 1996a).

Acid mine drainage contamination affects the benthic macroinvertebrate community in two ways. Either directly, by physiologically effects; and indirectly, caused by physical effects. The physiological effects result from the toxicity of the metals and low pH. Physical effects are caused by the precipitation of metals, particularly iron, as the pH in streams increases due to dilution or buffering properties. Iron precipitation can create a reduction in habitat heterogeneity in the substrate, and may also smother periphyton production (Harding et al., 2000; Penny, 1987).

In the Brunner Coal Measures, iron concentrations commonly become elevated in streams impacted by acid mine drainage, leading to precipitation and the armouring of substrate. Iron precipitate can also clog the mid-gut of invertebrates, and restrict or entirely prevent the absorption of food (Herrman et al., 1993). Furthermore, it can physically obstruct the gills of some invertebrates and reduce oxygen uptake from water which may already have reduced levels of dissolved oxygen (Quinn et al., 1992). The resulting physiological consequences of reduced gas-exchange and food-uptake efficiency may partially explain an inverse relationship between body-size and survivorship due to metal contamination in stream invertebrates (Kiffney & Clements, 1996b). Herrmann et al (1993) postulated that mayflies moult more frequently in mine-impacted waters in an effort to counteract the build up of iron precipitates on the cuticle, therefore alleviating hindered gill and gut function.

Iron precipitate can interfere with the metabolic processes and osmoregulation of invertebrates. At the cellular level, iron oxides can induce the formation of cell degenerating hydroxyl free-radicals (Vuori, 1995).

Furthermore, in addition to the toxicity associated with excessive levels of metals, metal compounds have high chemical oxygen demands, which may reduce the levels of dissolved oxygen in the water, placing populations under increased stress (Nimmo et al., 1998). Dissolved oxygen concentrations directly influence the abundance and diversity of invertebrates in freshwater communities (Boulton et al., 1997). Invertebrates that have been chronically exposed to metal contamination are also less likely to be tolerant of additional stresses such as food deprivation or insecticide application (Stone et al., 2001).

Streams that have high levels of dissolved metals and metal contaminated sediments typically show reduced invertebrate taxonomic richness (DeNicola & Stapleton, 2002; Jones et al., 1999). While total abundance is frequently low in comparison with reference sites, some studies of areas that are dominated by metal tolerant taxa like chironomids do not show a reduction in abundance, only a decrease in diversity (Richardson & Kiffney, 2000).

The response of freshwater invertebrate species to high concentrations of metals inevitably leads to population changes, such as reduced abundance, recruitment failure, and eventually the local extinction of sensitive species from the ecosystem.

Sub-lethal exposure to metals can reduce growth, survival and fecundity in aquatic invertebrates, causing a selection pressure for detoxification mechanisms, such as the production of metal-binding proteins, to arise in populations chronically exposed to contaminants (Vuori, 1994). This occurs through either phenotypic plasticity or natural selection. If the stream condition improves, benthic invertebrate populations that have been exposed and have adapted to high metal concentrations may be inferior competitors to individuals that arrive from naïve populations without such energetically expensive adaptations (Clements, 1999).

Changes in water chemistry associated with acid mine drainage may also lead to behavioural changes in benthic invertebrates. The combination of acidification and increased metal ion concentration can reduce feeding behaviour, and general activity, in mayflies (Ephemeroptera). For example, elevated cadmium concentrations have caused escape behaviours in the mayfly *Leptophlebia marginata*, resulting in an increase in passive drift, and subsequently increased mortality due to predation (Herrman et al., 1993). This is accentuated in streams that have pulsed contamination by heavy metals, for example those polluted by storm water run-off or industrial sewerage, or mine drainages that predominantly consist of surface water inputs as opposed to ground water inputs (Karouna-Renier & Sparling, 2001). Mayflies showed increased drift during periods of high metal concentrations, and it was during this time that stonefly predation had the greatest impact on mayfly populations (Clements, 1999; Kiffney, 1996; Merrett et al., 1991). This drives change in the community composition of polluted waters, as the most sensitive taxa, such as mayflies, disappear from the system first, through the accumulative effects of toxicity, predation, emigration and competition for potentially limited food sources.

Low pH conditions increase the toxicity of metals, as acidity makes metals more soluble, and thus more readily absorbed (Herrman et al., 1993). Furthermore, long-term exposure to metals can cause freshwater invertebrate communities to become more sensitive to low pH levels but more tolerant to metal contamination (Courtney & Clements, 2000).

The effects of metals and low pH are difficult to separate, because they seldom occur in isolation and have the synergetic consequences discussed above. From a remediation perspective it can be difficult and expensive to address both issues separately.

The aims of this investigation were:

1. To use experimentation to establish which was the dominant cause of toxicity; low pH, or high metal concentrations by manipulating pH in an attempt to isolate separate effects.
2. To investigate the toxicity of iron-armoured substrate to enable post-remediation predictions about the habitability of acid mine drainage (AMD) impacted streams.

It was logistically preferable to run these investigations in Christchurch where full laboratory facilities were available, and to use local invertebrates. Therefore, a validation experiment was conducted to ascertain if local invertebrates from a Banks Peninsula stream responded to experimental conditions in a fashion similar to that of West Coast invertebrate populations local to AMD areas.

METHODS

Three 96-hour experiments were conducted to compare the response of macroinvertebrates from a Banks Peninsula stream (Charteris Bay) and a Denniston Plateau stream to experimental conditions, and to investigate direct AMD toxicity to macroinvertebrates.

Mortality of the common leptophlebid mayfly, *Deleatidium,* was used as the response variable for these experiments. *Deleatidium* was selected as a study organism as it was easily maintained in the laboratory, it is abundant, and easily collected. Furthermore, *Deleatidium* seems to exhibit a tolerance to pH range that encompasses many other common benthic invertebrates (Harding et al., 2000). *Deleatidium* were collected from Charteris Bay Stream, Banks Peninsula, where this mayfly is abundant. The stream is relatively stable and easily accessed, and the animals could be collected and transported to the laboratory within two hours. However, Charteris Bay stream is circum-neutral, whereas many streams in the Buller region have a naturally low pH (eg pH 4.5). This may have resulted in a level of adaptation to low pH conditions in West Coast invertebrate populations. Although it would have been ideal to use animals from populations local to AMD sites, it is not feasible as mayflies are difficult to transport, and do not occur in high densities in this area. To examine this possibility a single "calibration experiment" was conducted, in which the response to experimental conditions of *Deleatidium* populations from Charteris Bay, and from a naturally low pH (pH 4.5) tributary to Rapid Creek (Denniston Plateau), were compared.

Experiment 3.1 Establishing if Banks Peninsula mayflies respond the same as West Coast mayflies.

The response of invertebrates from Canterbury to experimental conditions was compared with that of West Coast invertebrates.

Deleatidium mayflies were collected from Charteris Bay Stream, Banks Peninsula, and a small tributary to Rapid Creek (arbitrarily named Sullivans Road stream), Denniston Plateau. This tributary was not impacted by AMD, but had a pH of 4.5.

Animals were kept cool in insulated containers and transported to the laboratory within five hours of collection. Mayflies were left to acclimatise in large aerated buckets of stream water at 4°C for 24 hours. The stream water was sourced from the animals' stream of origin.

Deleatidium were assigned to dishes in a randomised blocking design to allow for statistical compensation of the variable sizes of study organisms.

This experiment was conducted in a controlled environment with an ambient temperature of 4.5°C, and a day/night light regime of 12:12 hours.

Each treatment was replicated five times. Each replicate consisted of a single covered plastic acid-washed petri dish, containing 5-10 animals, and 40 mls of water. Water in the dishes was oxygenated by continual aeration using compressed air. Throughout the exposure the volume in each experimental vessel was maintained at 40 mls (Figure 3.1).

The three water treatments were distilled water (pH 5.5), naturally acidic water from the Sullivans Road stream (pH 4.5), and AMD impacted water from the Rapid Creek main stem (pH 2.8).

Mortality was monitored every 24 hours for four days. Individuals were prodded with a blunt probe, and if they failed to exhibit a response they were considered dead.

Statistical Analysis

Results at 96 hours were arc-sine/square-root transformed and analysed using generalised linear model (GLM). The SAS System for Windows, Version 8 was used for this analysis.

Experiment 3.2 Distinguishing toxic effects of pH and metals

This experiment was conducted with the same regime as Experiment 3.1

The pH of each treatment was manipulated using 1M HCl and 1M NaOH. Following pH manipulation, each treatment was passed through Grade 1 filter paper, to extract any precipitates that formed due to the change in pH.

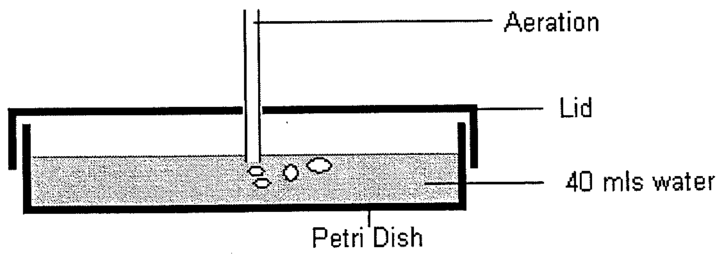


Figure 3.1 **Apparatus configuration for pH manipulation experiment.**

Water from Rapid Creek, collected downstream of the Sullivan Mine adits, represented AMD impacted water (containing dissolved metals), and distilled water was used as a control (not containing dissolved metals). Specific treatment details are outlined in Table 3.1.

Table 3.1 **Treatments used in the pH manipulation experiment.**

Treatment code	Water type	Initial pH	Adjusted pH
RCD2.8	Rapid Creek	2.8	Control (not adjusted)
RCD3.5	Rapid Creek	2.8	3.5
RCD4.0	Rapid Creek	2.8	4.0
RCD4.5	Rapid Creek	2.8	4.5
DW3.5	Distilled Water	5.5	3.5
DW4.0	Distilled Water	5.5	4.0
DW4.5	Distilled Water	5.5	4.5
DW	Distilled Water	5.5	Control (not adjusted)

Mortality was monitored every 24 hours for four days. Individuals were gently prodded with a blunt probe, and if they failed to respond they were considered dead.

Initial (0 hours) and final (96 hours) pH, conductivity and sulphate concentration were measured. pH and conductivity were measured using an Oaktron® Waterproof pH/Con 10 Meter. The concentration of sulphate ions was measured by collecting a water sample in an acid-washed polyethylene bottle, and analysed using a HACH DR2000 Direct Reading Spectrophotometer according to the method described in Hach (1992).

Samples of the Rapid Creek water following manipulation were collected and analysed for concentrations of total iron, dissolved iron, dissolved aluminium, dissolved nickel and dissolved zinc. Analysis was conducted by Inductively Coupled Plasma mass spectrophotometry by an accredited laboratory (RJ Hill Laboratories Limited, Hamilton).

The manipulated distilled water treatments were not analysed for metals, as they were not expected to contain significant amounts of metal ions.

Statistical Analysis

Results at 96 hours were arc sine, and square root transformed to normalise the data, and then analysed with analysis of variance (ANOVA). Significant differences in mortality were identified at $p<0.05$. An LSD test (post-hoc) was used to ascertain which specific treatments were different. The SAS System for Windows, Version 8 was used for this analysis.

Experiment 3.3 The toxicity of iron precipitate coated substrate

This experiment was designed to investigate whether iron precipitate contributes to changes in water chemistry, therefore resulting in toxic effects to macroinvertebrates.

The experiment was conducted in a controlled environment with an ambient temperature of 10°C, and a day/night light regime of 12:12 hours. Two treatments were replicated five times. Each replicate consisted of a single two litre plastic acid-washed container (ice-cream containers). An assortment of substrate particles (gravel and small cobbles) was placed in the bottom of each container, along with one litre of water. Water was continually aerated for the duration of the experiment (Figure 3.2). Ten *Deleatidium* were randomly allocated to each container. Treatments are detailed in Table 3.2.

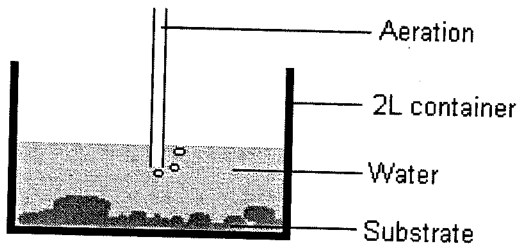


Figure 3.2 **Apparatus configuration for substrate experiment.**

Water was collected from Rapid Creek, upstream of the Sullivans Mine adits. Previous experimentation has shown that although this water has a low pH (4.5), it did not cause significant mortality to *Deleatidium* (K.O'Halloran, unpublished data).

Substrate was collected from the Rapid Creek stream bank from the dry sides of the main channel in an attempt to minimise variability due to periphyton present on the substrate.

Mortality was assessed every 24 hours for four days as in Experiments 3.1 and 3.2.

Conductivity, pH and sulphate concentration were measured at the commencement and conclusion of the experiment. pH and conductivity were measured using an Oaktron® Waterproof pH/Con 10 Meter. The concentration of sulphate ions was measured by collecting a water sample in an acid-washed polyethylene bottle, and analysed using a HACH DR2000 Direct Reading Spectrophotometer according to the method described in Hach (1992).

Table 3.2 **Experimental treatments investigating the toxicity of iron precipitate.**

Treatment code	Substrate origin	Iron precipitate	Water origin
RCU-D	Rapid Creek, downstream of Sullivan Mine adits	Present	Rapid Creek, upstream of Sullivan Mine adits
RCU-U	Rapid Creek, upstream of Sullivan Mine adits	Absent Control treatment	Rapid Creek, upstream of Sullivan Mine adits

Statistical Analysis

Results at 96 hours were arc sine, and square root transformed to normalise the data, and analysed using a paired t-test (Microsoft ® Excel 2002).

RESULTS

Experiment 3.1 Comparing the response of Banks Peninsula mayflies and West Coast mayflies.

Initially there was some variation in the response of mayflies from Banks Peninsula and the West Coast when exposed to the experimental treatments,. Charteris Bay (Banks Peninsula) mayflies were initially tolerant of AMD impacted water from Rapid Creek, whereas 80% of animals from Sullivans Road stream experienced mortality within 24 hours (Figure 3.3). There was no significant difference in the response of the two populations to water from Sullivans Road stream or distilled water under experimental conditions at any time throughout the duration of this experiment.

After 96 hours there was no difference in the response of the two populations to any of the water types. The GLM procedure indicated that the only significant determining factor in mayfly survival was the type of water used in the treatment (Table 3.3). There was no significant difference in survival due to the origin of the animals used.

Given this result, only data recorded at 96 hours was used for analysis in further toxicology experiments.

Table 3.3 Results of the GLM analysing the response of West Coast (Denniston Plateau), and East Coast (Banks Peninsula) mayflies to experimental treatments.

	DF	Mean square	F	p
Block	4	0.02	0.34	0.85
Water type (AMD, naturally low pH, distilled water)	2	5.65	120.57	0.0001
Mayfly type (West Coast or Banks Peninsula)	1	0.17	3.73	0.07
Water type x Mayfly type	2	0.004	0.08	0.92
Error	20	0.05		
Total	29			

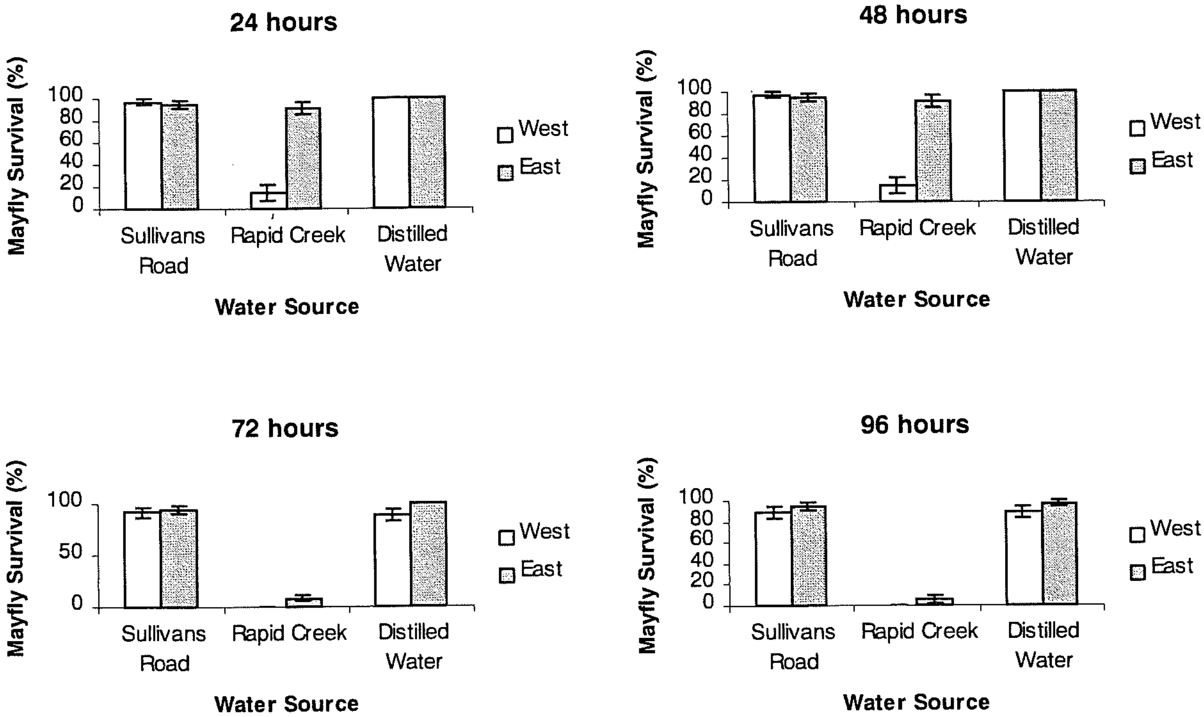


Figure 3.3 **Mean survival of *Deleatidium* mayflies (±SE) at each time period in Sullivans Road water (naturally low pH), Rapid Creek water (AMD impacted), and distilled water. Grey bars signify Charteris Bay mayflies (East); white bars Sullivans Road mayflies (West).**

Experiment 3.2 Distinguishing toxic effects of pH and metals

Manipulating the pH of Rapid Creek water considerably altered its water chemistry. Figure 3.4 shows changes to the conductivity of Rapid Creek water and distilled water, following pH manipulation. The conductivity of Rapid Creek water was lower in the manipulated treatments. The manipulation of the pH of distilled water resulted in a slight increase in conductivity.

The concentration of iron, aluminium and sulphate ions also changed as the pH of Rapid Creek water was artificially increased (Figure 3.5). The pH manipulation caused metals to precipitate, and the filtration process removed these. Sulphate, total iron and dissolved aluminium showed gradual reductions in concentration, whereas dissolved iron was more abrupt. When the pH was adjusted to 3.5, 98% of dissolved iron present in Rapid Creek samples prior to manipulation (pH 2.8) was removed.

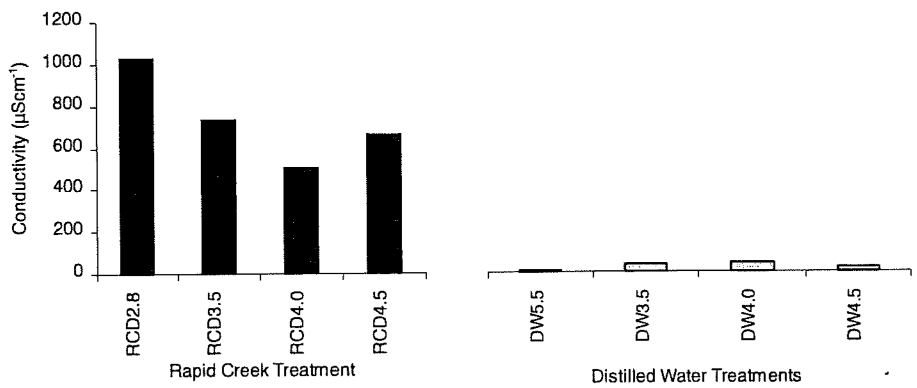


Figure 3.4 The conductivity (μScm^{-1}) of the treatment waters. RCD signifies Rapid Creek water (downstream of AMD impacts), and DW represents distilled water treatments. RCD2.8 indicates the initial conductivity of Rapid Creek water prior to pH manipulation, and DW5.5 indicates the conductivity of distilled water prior to manipulation. For a full explanation of codes see Table 3.1.

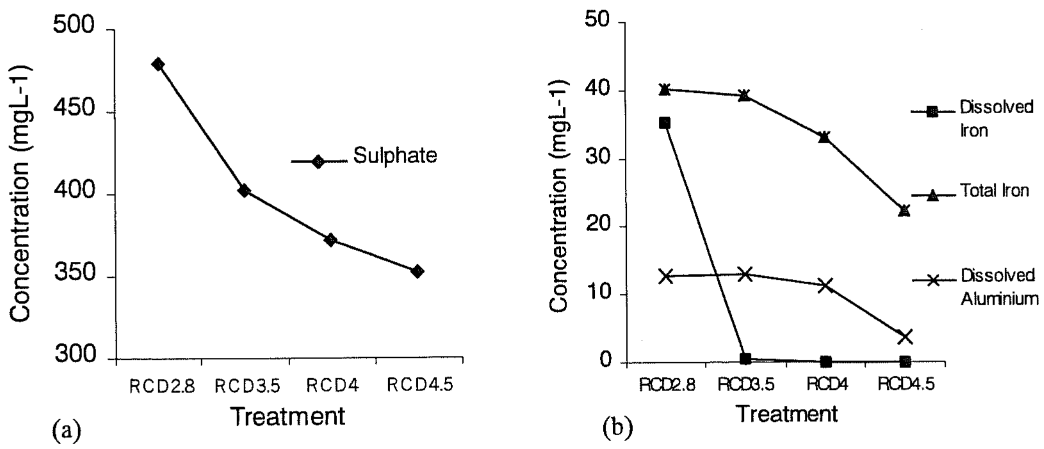


Figure 3.5 **Ion concentrations in Rapid Creek water after manipulation to various levels of pH. RCD2.8 represents Rapid Creek water prior to any manipulation. (a) Sulphate ion concentrations in Rapid Creek water after manipulation of pH, and (b) Iron (total and dissolved) and aluminium concentrations in Rapid Creek water after manipulation of pH.**

There was no significant difference in the response of *Deleatidium* to any of the treatments with pH levels of 4.0 or 4.5 (Figure 3.6). The ANOVA indicated that pH was the only significant factor in determining mortality (Table 3.4). The variety of water, distilled or Rapid Creek, did not significantly affect the model. The RCD3.5 treatment had considerably higher levels of metal ions than the other manipulated treatments (Figure 3.5).

Table 3.4 **Results of the ANOVA analysing the response of mayflies to pH adjusted AMD and distilled water.**

	DF	Mean square	F	p
Block	4	0.033	0.36	0.84
pH	3	1.025	16.60	0.0001
Water type (AMD or distilled)	1	0.224	3.62	0.07
pH x water type	2	0.011	0.18	0.83
Error	24	0.062		
Total	34			

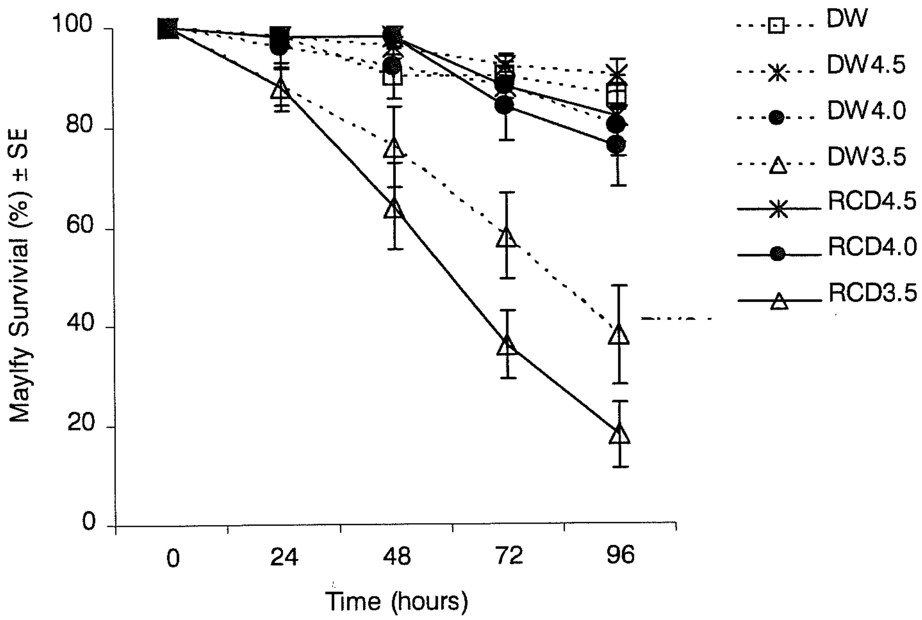


Figure 3.6 **Mean *Deleatidium* survival (±SE) in pH manipulation treatments at each 24-hour mortality assessment period for 96 hours. DW; distilled water treatments and RCD, Rapid Creek water treatments, with pH of each treatment after manipulation.**

Experiment 3.3 The toxicity of iron precipitate on substrate

The chemistry of the water from Rapid Creek, upstream of AMD inputs, exhibited some changes over 96 hours when used in treatments with substrate that did or did not have iron precipitate. These changes were minor in the treatment without iron-precipitate (RCU-SU) in comparison to the treatment where iron precipitate was present (RCU-SD). The water in the treatment with iron precipitate decreased in pH by more than two units and increased in conductivity by 150μS (Figure 3.7). Similarly, the concentration of sulphate ions increased considerably when iron precipitate was introduced.

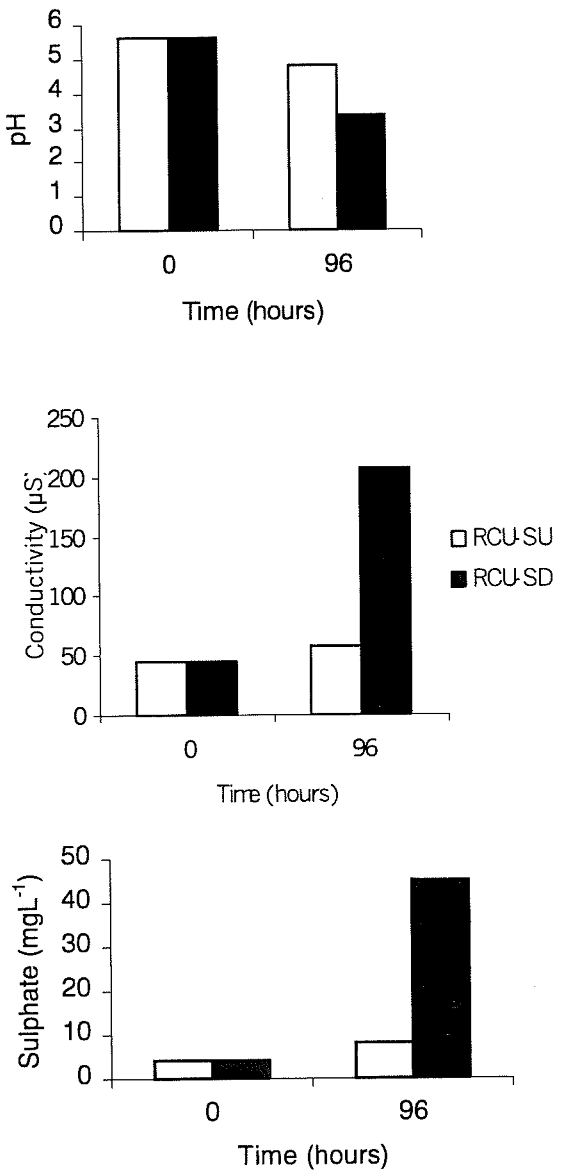


Figure 3.7 **Water chemistry of the substrate experiment over 96 hours. RCU-SU is substrate lacking iron precipitate with water from Rapid Creek Water, upstream of Sullivan's mine (naturally low pH but not impacted by AMD) and normal substrate, and RCU-SD is the same water, but with substrate that is coated in iron precipitate. Top: pH, middle: conductivity, bottom: sulphate ion concentration.**

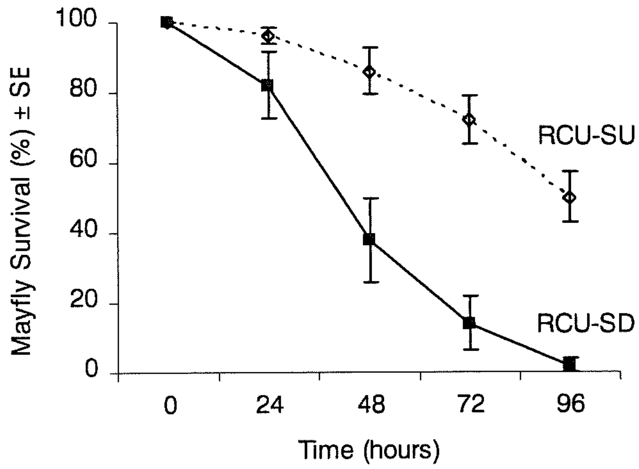


Figure 3.8 Mean survival (±SE) of *Deleatidium* over 96 hours when exposed to the two substrate types. RCU-SU is pre-AMD Rapid Creek water and iron-free substrate, and RCU-SD is pre-AMD Rapid Creek water and iron-armoured substrate.

Deleatidium in both treatments suffered mortality after 96 hours (Figure 3.8). However, survival was significantly lower in the treatment that incorporated iron-armoured substrate (RCU-SD), as only a mean of 2% of individuals survived, compared with a mean 50% in the control treatment (T_4=9.20, p=0.0008).

DISCUSSION

Mayflies from Charteris Bay stream (Banks Peninsula) showed very similar responses to AMD water and experimental conditions to mayflies from Sullivans Road stream (Denniston Plateau). The population from the circum-neutral stream (Charteris Bay) were able to tolerate the toxicity of Rapid Creek water for slightly longer than their counterparts from a naturally acidic West Coast stream. This may be due to a number of factors. For example, animals from the Charteris Bay stream may have more opportunity to feed, as the stream is more stable than those on the Denniston Plateau. Therefore, these mayflies may have a higher quality food source, and resulting in them being in better physical condition. Alternatively, the animals may have responded differently because of the additional stress placed on mayflies from the West Coast, which were subject to a prolonged period of transportation.

However, despite initial disparities, after exposure to the water treatments for 96 hours there was no difference in mortality between animals from the two streams. This indicates that it was acceptable to use animals from the Charteris Bay stream population for further 96-hour toxicity tests, and it was anticipated that the response of Banks Peninsula mayflies to AMD treatments might be similar to the responses of mayflies from West Coast streams with naturally low pH levels.

A clear distinction between the affect of toxic metals and low pH was difficult to detect in these manipulations because significant quantities of the metals present in the AMD water were removed (>50%) when the pH was manipulated. Furthermore, pH and metal concentrations are generally correlated in natural conditions, and may even have synergetic toxic effects for invertebrates (Merrett et al., 1991). This 96-hour experimental procedure does not account for these complex relationships, but it does give some indication as to important mechanisms of the toxicity of AMD.

Results indicate that aluminium and iron remained in the manipulated treatments at reduced concentrations, therefore providing a comparison with the distilled water treatments that had altered pH levels and no metals.

These results suggests that water with a pH of 3.5 is toxic for *Deleatidium* regardless of whether or not metals are present, although toxicity may be increased by the presence of metals. When pH is at 4 or higher, AMD water is much less toxic, and any metals present do not appear to have additional detrimental affects.

Allard and Moreau (1986) had similar findings with experimental acidification and addition of aluminium to artificial stream channels that suggested that lowering the pH of stream water had a significant impact on the density of individuals in the invertebrate community, and reduced colonisation. However, the addition of aluminium, even in concentrations as high as 30 mgL^{-1}, did not adversely affect macroinvertebrates. Furthermore, Dills and Rogers (1974) found that some chironomids, megalopterans, and ceratopogonids were surviving in acid mine drainages with high metal concentrations. These individuals had absorbed significant amounts of metal ions without any apparent detrimental affects. It is likely that there is variation between insect groups in their ability to physiologically or morphologically exclude such metals (Winterbourn et al., 2000). Parsons (1977) suggested that some species were becoming adapted to acid mine effluent, due to some streams being subjected to a continuous flow. The selection pressure applied to chironomids by metal contamination resulted in the evolution of increasingly dynamic life-history patterns, suggesting that adaptation to the presence of metal toxicants was occurring (Groenendijk et al., 1999).

As pH levels increase and are no longer toxic, benthic communities are still impacted by the effects of AMD. At less toxic levels of pH the toxicity of dissolved metals in the water column, and metals impregnated in the sediment, continue to affect the structure of macroinvertebrate communities (Schmidt et al., 2002). Invertebrate communities in the co-occurring naturally acidic streams (low pH, low metal concentrations) and AMD-impacted streams (low pH, high metal concentrations) on the West Coast are very different (Collier et al., 1990). AMD impacted streams generally have lower macroinvertebrate diversity than naturally acidic streams, and communities are frequently dominated by Dipteran taxa (Winterbourn & McDiffett, 1996). These differences are almost certainly due at least in part to the elevated metals in AMD impacted streams (Collier et al., 1990).

The iron precipitate on the substrate of AMD impacted streams is a further factor that is likely to be affecting benthic invertebrates (Cherry et al., 2001). Precipitated iron flocs on the substrate of AMD impacted streams have been thought to be limiting macroinvertebrates chiefly due to a series of physical effects on the habitat in these streams. This precipitate is considered to smoother periphyton, limiting food availability for macroinvertebrates (Harding et al., 2000). Iron-depositing bacteria restrict diatom growth, while being unsuitable as a food source in themselves

(Wellnitz et al., 1994). Furthermore, the consolidation of the sludgy floc in the streambeds blocks interstitial spaces, and therefore limiting the availability of refugia for invertebrates during high flows. Many invertebrate taxa rely on these refugia to resist disturbance events caused by flooding (Holomuzki & Biggs, 2000), which is a frequent phenomenon in the Buller Region.

This study shows that in addition to altering the physical environment, iron oxide-coated substrate can alter water chemistry. This resulted in significant increases in *Deleatidium* mortality. The control treatment had some mortality after 96 hours, suggesting the experimental procedure was stressful for the mayflies. However, the iron precipitate caused a marked reduction in pH, which resulted in significantly more mortality.

Furthermore, regardless of water chemistry, iron oxide precipitates and other sediments produced by AMD can continue to be toxic. Aeration of AMD sediments can lead to a reduction in ambient pH, which in turn results in the mobilisation of other metals such as zinc and manganese (Fajtl et al., 2002). As iron precipitate can remain on substrate particles for a substantial period after transferral into circum-neutral water (Niyogi et al., 1999), iron oxide precipitates are likely to continue to affect the macroinvertebrate community of AMD impacted streams even if water chemistry is adequately remediated.

Iron armouring can be caused both by chemical and biological processes. Blooms of the bacteria associated with depositing iron floc can occur in certain conditions, particularly when ambient iron concentrations are high. These bacterial blooms can have many detrimental affects on macroinvertebrates (Wellnitz et al., 1994). These include food limitation, and direct toxic affects. Iron-depositing bacteria can also produce avoidance behaviour of iron-armoured substrate in many invertebrate taxa (Wellnitz et al., 1994).

The implication of this result for remediation attempts is that although simply addressing the issue of pH will be sufficient to render the water chemistry more tolerable for macroinvertebrates, the retention of iron oxide on stream substrates may result in the water chemistry remaining toxic, and therefore inhibiting any recovery of the benthic community.

Although the contamination of freshwater ecosystems by metals can have a profound impact on the invertebrate community, pH seems to be the most important factor causing toxicity in AMD water. Metals can directly affect invertebrates by causing malformations, behavioural changes and even mortality at elevated concentrations. This can lead to morphological and even genetic changes that may be disadvantageous to a population should the condition of the environment change. Indirect effects of metal pollution can include food limitation as periphyton growth is suppressed or made toxic, and the elimination or emigration of sensitive taxa, resulting in alterations in community composition and structure. However, when pH is at tolerable levels, invertebrates seem much better able to tolerate the sub-lethal affects of metal contamination.

Iron-armoured substrate is likely to remain a limiting factor for macroinvertebrates in remediated streams, as chemical and biological processes associated with it can contribute toxic compounds to the water column, while it continues to physically render the habitat inhospitable. However, West Coast streams, particularly in the Denniston Plateau area, typically have a very steep gradient, and are prone to frequent flood events. This may be an advantage for remediation, as the numerous disturbance events may create an element of abrasion, which will lead to a more rapid removal of the iron precipitate than might be observed under more benign conditions.

Chapter Four

Iron precipitate on substrate: effects on organic biofilm and benthic invertebrates

Streams with acid mine inputs typically have high concentrations of dissolved metals in the water column. This is characteristic of systems associated with the Brunner Coal Measures where high sulphate and metals are associated with the geology (Winterbourn et al., 2000b). The process of coal extraction increases oxidation of iron pyrite, resulting in high concentrations of iron oxides being released. The lowered pH can also mobilises other metals such as aluminium, arsenic and zinc (Niyogi et al., 2001). The precipitation of iron results in a flocculation effect, which can completely smother the substrate (Letterman & Mitsch, 1978). Aluminium hydroxide can also form a precipitate, but is less common on the West Coast because of the naturally low pH levels of the area.

Iron precipitate directly affects macroinvertebrates, both physically and physiologically. Physically, it clogs respiratory surfaces and gut linings (Mason, 1996), and physiologically it is chemically active, with the potential to be lethal to sensitive taxa (refer Chapter 3).

In addition to toxic and physical effects, iron precipitate may limit periphyton growth. This may occur because fine precipitate particles may increase the turbidity in the water column and inhibit photosynthesis (Davies-Colley et al., 1992), and also because the iron hydoxide may directly smother periphyton growth (Sode, 1983). Reduced periphyton growth may limit food availability for invertebrates (Wellnitz & Sheldon, 1995). Furthermore, algae and other micro-organisms that can live in association with iron precipitate-coated substrate can accumulate toxic compounds. For example, mayfly nymphs were unable to survive when experimentally fed algae harvested from low pH, metal contaminated streams (Herrman et al., 1993). Bryophytes also accumulate toxic compounds, and high moss biomass can lead to a high incidence of morphological abnormalities in aquatic invertebrates in metal contaminated streams if there is not an alternative food source (Vuori & Parkko, 1996). In polluted streams herbivorous invertebrates that are partially food limited, have been shown to change their energy investment regime to invest more in detoxification mechanisms, and therefore had fewer resources to invest in growth and reproduction (Vuori, 1994).

Water acidified by mine drainage often supports dense growths of acid-tolerant filamentous algae like *Ulothrix* sp. (Niyogi et al., 1999). Experimentally acidified channels had higher concentrations of chlorophyll a, because of this acidophilic alga, than circum-neutral channels (Allard & Moreau, 1986). However, the filamentous morphology of algae such as *Ulothrix* sp. is likely to limit its usefulness as a food source for New Zealand macroinvertebrate grazers, which prefer diatom algae as a food source (Suren et al, 2003).

Furthermore, increased deposition of iron precipitates limits the growth of *Ulothrix*. In addition to iron precipitate being limiting, Niyogi et al (1999) showed that at higher levels of pH (>5) dissolved aluminium hydroxides will precipitate, and this can entirely eliminate growth of all periphyton, including filamentous algae (Niyogi et al., 1999).

The cumulative effect of this food limitation will impact the entire food chain if herbivore growth is restricted (Elser et al., 2000), and will eventually impact on predators, unless they are subsidised from other environments, or can effectively prey-switch to less sensitive taxa.

An additional indirect effect of high concentrations of iron in the water causes turbidity that limits the growth of epilithic periphyton like diatoms (because of their requirement for light), while at the same time it elevates the growth of ferromaganese-depositing bacteria (Wellnitz & Sheldon, 1995). These bacteria can further accelerate iron precipitating on the substrate surface. This can lead to a positive feedback cycle whereby blooms of ferromaganese-depositing bacteria exclude other micro-organisms, reducing competition for resources and therefore facilitating more bacterial growth. This is likely to have a cumulative effect on benthic macroinvertebrates.

Firstly, diatoms are an important food source, and the absence of this is likely to have effects throughout the trophic levels. Macroinvertebrates are unlikely to be able to use the ferromaganese-depositing bacteria themselves as a food source (Wellnitz et al., 1994).

Secondly, the bacteria themselves are hetertrophic and are therefore direct competitors for resources in the water column such as oxygen and nutrients. Blooms of ferromaganese-depositing bacteria appear to be toxic to macroinvertebrates, and also can cause altered behaviour (Wellnitz et al., 1994). Macroinvertebrates tend to actively avoid surfaces coated in blooms of ferromaganese-depositing bacteria (Wellnitz et al., 1994).

The combination of the toxic, physical and indirect affects of iron precipitate on benthic macroinvertebrates is likely to be affecting the community composition in acid mine drainage-impacted streams.

The aims of this study were:

1. To establish if AMD substrate has reduced availability of organic biofilms. I hypothesise that the presence of iron hydroxide precipitate on the substrate will limit the growth of epilithic algae.
2. To investigate if macroinvertebrates actively avoid substrate coated in iron oxide precipitate.
   I hypothesise that initially macroinvertebrates will avoid substrate coated with iron-hydoxide precipitate, but that with increased time immersed in non-AMD water and the precipitate re-dissolves this avoidance will become less apparent.

Streambed substrate in mine impacted catchments can have highly variable levels of iron precipitate. This provides an opportunity to investigate if different amounts of precipitate differentially affect the behaviour and colonisation of macroinvertebrates.

Rapid Creek (see Chapter 1) is a small stream with little dilution of the AMD water that flows into it. This results in the substrate being heavily coated in iron-hydroxide precipitate. Contrastingly, substrate in the Waimangaroa River is only moderately coated. This is probably due to the fact that the Waimangaroa River is a much larger catchment, has greater discharge, and thus greater dilution of the AMD inputs. There is also likely to be a higher level of mechanical abrasion in the lower reaches of this river, as the bed is wide and braided.

METHODS

Field Survey

A field survey of AMD impacted catchments and control (non-AMD impacted) catchments was conducted, and quantitative data on organic biofilm biomass and macroinvertebrate communities were obtained. The biological community was quantified by collecting three Surber samples (0.1 m^2; 0.5mm mesh). Where possible, samples were collected from random positions within riffle habitat. A further kick-net composite sample was collected of all microhabitats present at each site. Samples were preserved in 70% ethanol and returned to the laboratory for processing.

Macroinvertebrate samples were sieved through a 500μm mesh, sorted, enumerated and identified. Individuals were identified to genus when possible, and otherwise to the lowest feasible taxonomic level (Winterbourn et al., 2000a), magnified up to 35x using a binocular dissecting microscope. Chironomidae were mounted on slides with lactophenyl PVA and identified to genus by magnifying 100x – 400x using a stereo microscope (Winterbourn et al., 2000a),(Taylor, 2001).

Periphyton / organic biofilm biomass was quantified by collecting three replicate samples from each site. A known area of stone surface was scrapped carefully with a nylon brush, and the periphyton removed collected in a small quantity of stream water (Davies & Gee, 1993). Samples were stored in the dark on ice, and returned to the laboratory for analysis. Collected particulate matter was filtered out of the water using pre-ashed glass filters. These were then dried for 12 hours at 65 °C. The dry-weight of filters were measured, and then they were ashed for one hour at 450 °C. The ash-free dry mass of each sample was established, and averaged to give a mean mass of organic matter present per m^2 of substrate.

In-situ experiment

A substrate manipulation experiment was conducted in Grasmere Stream. Grasmere Stream is a stable, lake-fed, circum-neutral stream at Cass in the Arthur's Pass area.

Individual stones were used as experimental units. Ten stones were randomly selected from each of three streams (Rapid Creek, Waimangaroa River and Grasmere

Stream), representing different levels of iron precipitate (Table 4.1). Stones varied in size but were approximately 20cm x 10cm. Any animals attached to stones were removed prior to the experiment. Stones from Grasmere Stream were collected from outside the study reach and subject to the same conditions as stones from the other streams prior to the commencement of the study (i.e. placed in plastic bags as if for transportation).

Stones were placed in a riffle/run reach of Grasmere Stream in ten blocks of three. Each block had one stone from each stream. Blocks were randomly placed in Grasmere Stream within the 20-metre study reach.

After seven days all the stones from five blocks were collected using a net. Any macroinvertebrates that were dislodged when the stones were removed, attached to the stones or associated with individual stones were considered to have "selected' that stone as an appropriate habitat, and were therefore collected and preserved using 70% ethanol. Organic biofilm biomass was also quantified by establishing the ash-free dry mass of matter removed from a specific area of the surface of three stones (using the same methodology as in the field survey).

After four weeks this procedure was repeated with the remaining five blocks.

Macroinvertebrate samples were sieved through a 500μm mesh and sorted. Individuals were identified to genus when possible, and otherwise to the lowest feasible taxonomic level, using a binocular dissecting microscope (Winterbourn et al., 2000a). Chironomidae were mounted on slides with lactophenyl PVA and identified to genus at 100x – 400x using a stereo microscope (Winterbourn et al., 2000a),(Taylor, 2001).

Table 4.1 **Catchment origin of experimental stones.**

Stone	Stream/Catchment of origin	Level of iron precipitate
R	Rapid Creek	High
W	Waimangaroa River	Medium
G	Grasmere Stream	None

Statistical Analysis

A two-tailed t-test was used to compare the mean organic biofilm biomasses of AMD impacted and control catchments, using Microsoft ® Excel 2002. A correlation analysis of organic biofilm biomass and density of macroinvertebrates was performed using Microsoft ® Excel 2002. A correlation analysis organic biofilm biomass and macroinvertebrate diversity was also performed.

A one-way analysis of variance (ANOVA) was used to compared organic biofilm biomass means of the three types of stone prior to the commencement of the experiment, after one week, and after four weeks (Microsoft ® Excel 2002).

One-way analysis of variance (ANOVA) procedures were used to compared the number of taxa, the density of macroinvertebrates, the number of EPT taxa and the % EPT individuals in the community associated with the three substrate types after one week, and weeks (Microsoft ® Excel 2002).

RESULTS

Field Survey

AMD impacted sites had a higher biomass of organic biofilm on the substrate than reference sites (Figure 4.1). AMD impacted catchments had an average biomass of organic biofilm of more than 3 gm^{-2}, whereas reference catchments had less than 1 gm^{-2}. This was statistically significant difference ($T_{10}=-3.34$, $p<0.05$).

Quantity of organic biofilm is positively correlated with macroinvertebrate density, although this is only a weak correlation with a coefficient of $r = 0.37$ (Figure 4.2). Total taxa was negatively correlated with organic biofilm, with a coefficient of $r = -0.46$.

In-situ experiment

There no significant difference in the biomass of organic biofilm on the removed from the surface of rocks with the three different concentrations of iron precipitate (high, moderate, none) prior to the experiment commencing, or after one week ($p>0.05$). Similarly, after one month there was also no difference ($p>0.05$) (Table 4.2).

Table 4.2 **Mean organic biofilm (±SE) on each stone type, before the experiment began, after one week and after four weeks.**

Mean organic biofilm (±SE) (gm^{-2})	Level of iron precipitate: High	Moderate	Low
Initial	1.95 (±1.15)	0.57 (±0.16)	1.20 (±0.33)
1 week	1.60 (±0.16)	0.73 (±0.18)	1.18 (±0.32)
4 weeks	2.63 (±0.79)	3.39 (±2.76)	2.98 (±2.08)

After one week there were less macroinvertebrates associated with iron precipitate coated substrate than substrate that was not coated, although the ANOVA procedure indicates that this is not statistically significant. Accordingly, there was slightly a slightly higher biomass of organic biofilm collected from iron precipitate substrate (1.6 gm^{-2}, as opposed to 1.2 gm^{-2}), although again this was not statistically significant difference.

Similarly, no difference was found in the mean number of taxa present on stones in response to the amount of iron precipitate, nor was there any relationship between the amount of time (one week as compared with four weeks) that these stones have been in a circum-neutral stream, and the mean number of taxa.

Figure 4.3 shows that the number of taxa (diversity) associated with stones that were not covered in iron precipitate after one week was higher than the number associated with stones that were. However, the data was highly variable and was not statistically significant ($p>0.05$).

Similarly, the mean number of individuals per stone (abundance) appeared to be higher on stones free of iron precipitate (Figure 4.3), but again this was not significantly different ($p>0.05$).

After one month there is clearly no difference in the diversity or abundance of macroinvertebrates on any of the stone types.

Figure 4.4 shows that the number of EPT taxa associated with stones that were not coated in iron precipitate after one week was higher than the number associated with stones that were. However, the data again was highly variable and not statistically significant ($p>0.05$). Similarly, the proportion of EPT individuals in the community (% EPT) appeared to be higher on iron precipitate free stones (Figure 4.4), but again this was not a statistically significant difference ($p>0.05$).

After one month this trend is no longer apparent. % EPT is much higher on all three types of stone, but there is still no significant difference between them. The data is still highly variable, but the number of EPT taxa or % EPT shows no relationship to the degree of iron precipitate.

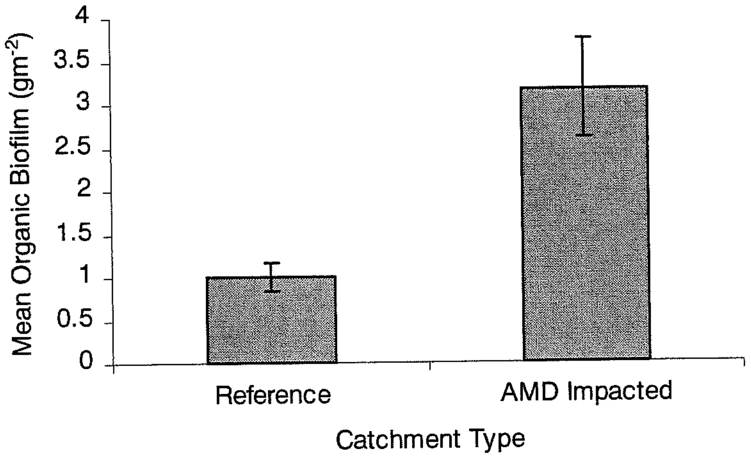
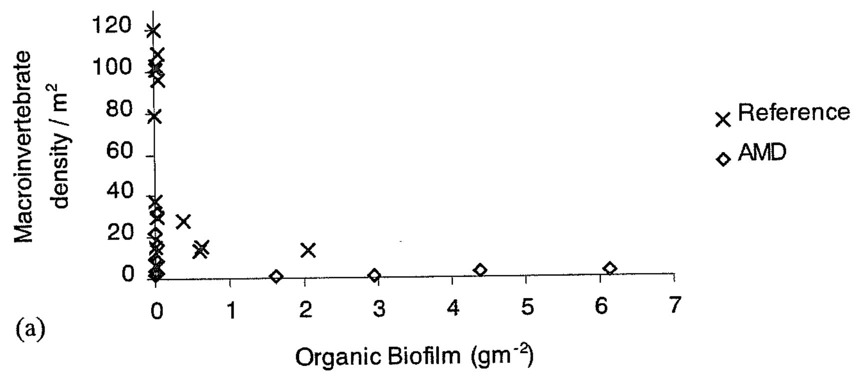
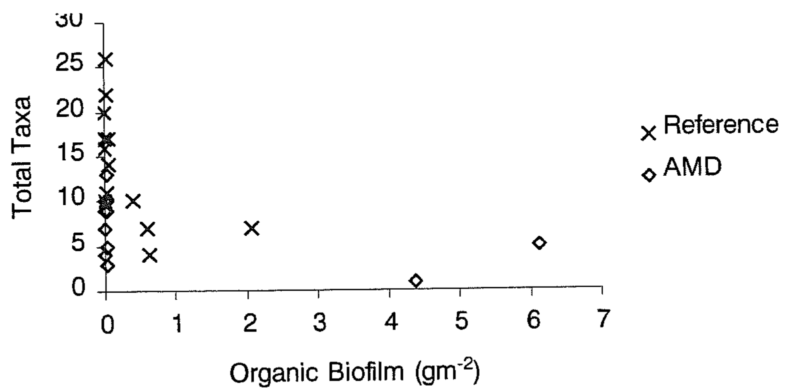


Figure 4.1 **Mean organic biofilm (±SE) on substrate surface from AMD impacted catchments and control catchments. NB no data was obtained from Charming Creek 5 as this site was entirely bedrock, and organic biofilm was not measured for Darcy Stream.**



(a)



(b)

Figure 4.2 **Correlation of ash-free dry mass of organic biofilm and (a) macroinvertebrate density , r = 0.37; and (b) total taxa, r = 0.46.**

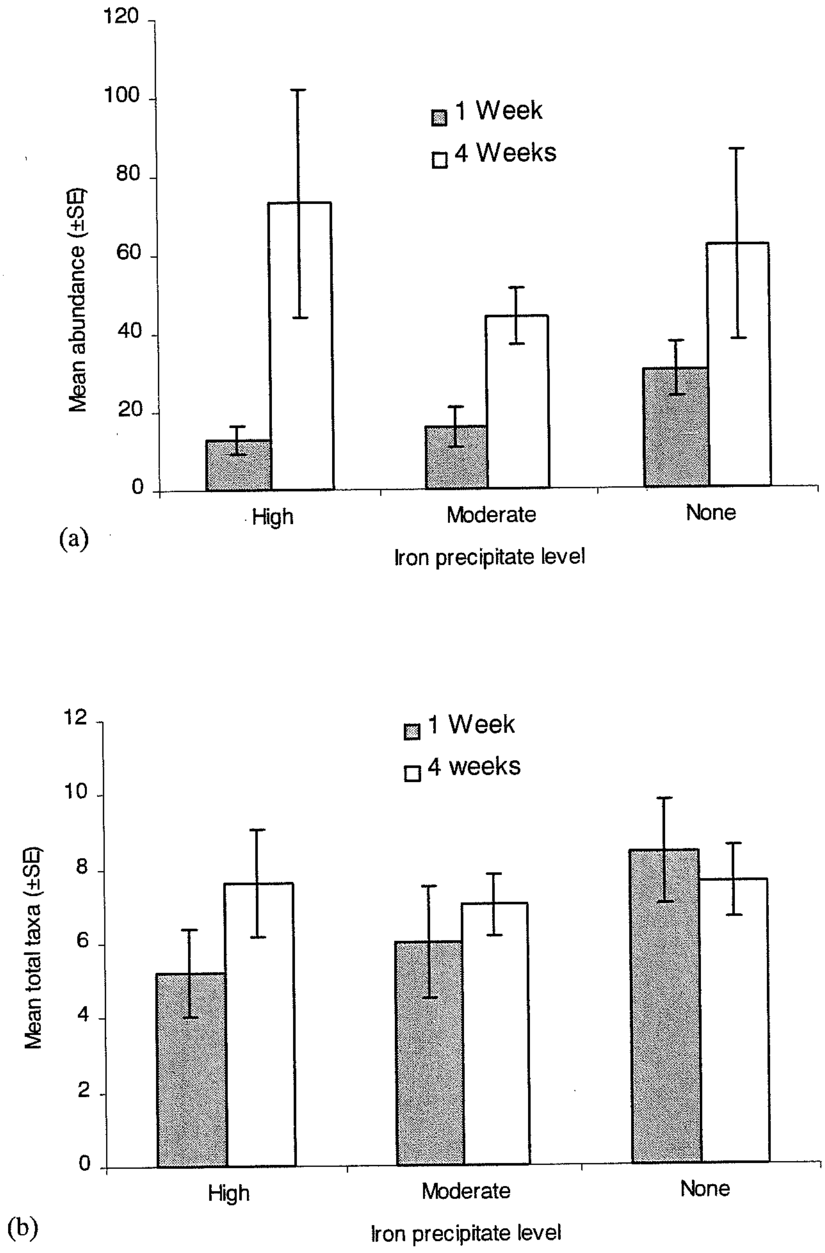
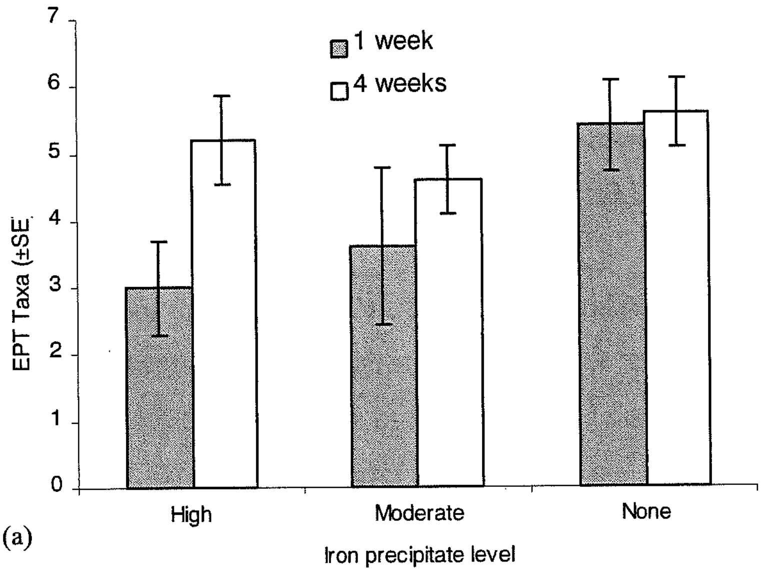
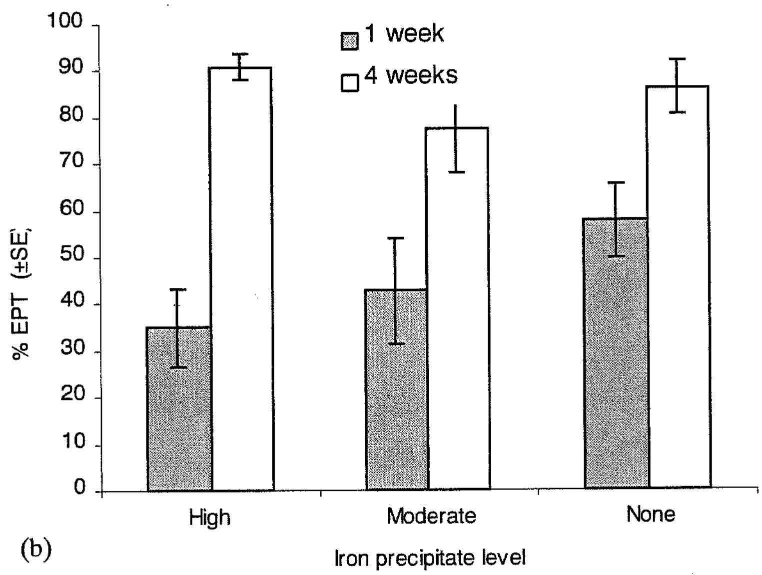


Figure 4.3 **Mean abundance (±SE) per stone(a) and mean total taxa (±SE) per stone for each level of iron armouring after one week (grey bars), and after four weeks (white bars).**



(a)



(b)

Figure 4.4 **Mean EPT taxa (±SE) per stone, (a), and mean % EPT (±SE) per stone (b), for each level of iron precipitate after one week (grey bars), and after four weeks (white bars).**

DISCUSSION

There was considerably higher biomass of organic biofilm growing on substrate in AMD impacted streams than in reference streams. Although this does not support the hypothesis that iron-hydroxide precipitates are smothering the growth of periphyton, it is probably explained by a reduction in grazing invertebrates in AMD impacted streams.

The positive correlation of macroinvertebrate density and organic biofilm biomass is likely to be due to a release from competition and predation for tolerant taxa like chironomids (Winterbourn, 1998). These taxa may increase in density as the quantity of organic biofilm increases. The negative correlation of taxa richness with organic biofilm further supports this idea. Highly diverse benthic communities generally include members of the trophic guild that graze epilithic algae (Suren et al, 2003). When this diversity is lost because of AMD contamination, so too are many links in the food web. AMD impacted streams have lower diversity and higher biomass of organic biofilm, which may be an indication that such streams have ecosystems that are reduced in complexity.

Harbrow (2001) found that AMD impacted streams on the West Coast commonly supported high densities of chironomid larvae (>500 individuals m^{-2}). These streams also typically had high algal biomass in comparison to streams with lower densities of animals, but greater diversity of taxa.

This study does not support the hypothesis that AMD impacted streams have reduced organic biofilm biomass due to smothering by iron-hydroxide precipitates. However, I did not distinguish between heterotrophic (e.g. ferromanganese depositing bacteria) and autotrophic (e.g. photosynthetic Chlorophyta) microbial taxa that may be components of the epilithic biofilm. Harbrow (2001) used a scanning electron microscope to show that in streams with high levels of iron hydroxide precipitate the epilithial communities tended to be dominated by bacteria, although often smothered diatoms were evident. Further research using an in-situ exclosure experiment would possibly provide a scientifically robust mechanism to further investigate the autochthonous productivity of AMD impacted streams and the effect of iron-hydroxide deposition on periphyton.

DeNicola and Stapleton (2002) carried out an in-situ experiment in AMD impacted streams, and found that iron precipitate on substrate was not associated with either the quantity of periphyton or invertebrate abundance or community composition.

Substrate with iron precipitate alters the ambient water chemistry under experimental conditions (refer Chapter 3). Furthermore, flushing or scouring action of water over precipitate or sediment impregnated with trace metals re-mobilises metal ions into the water column (Finlayson et al., 2000). It was hypothesised that when iron-precipitate coated substrate was introduced to Grasmere Stream trace metals would be released, producing a toxic layer at the interface between the water and the substrate. This would result in macroinvertebrates avoiding substrate with iron precipitate. My experiment did not find this, although there is an indication that there may be some avoidance of iron precipitate after a short period of time (1 week). The absence of a significant finding may be simply due to a lack of statistical power, and could possibly be improved with increased replication or a more robust experimental design (e.g. using large patches of substrate as experimental units as opposed to individual stones). However, it seems more likely that with such small quantities of iron precipitate substrate in Grasmere Stream the discharge of circumneutral water can adequately buffers the toxicity of the substrate. Further experimentation might also include using artificial channels in a laboratory environment. This would allow discharge to be controlled, and specific animals introduced, and their resulting behaviour monitored to detect any specific response to iron precipitate.

This investigation did not support the hypothesis that benthic macroinvertebrates actively select against substrate with iron precipitate in circum-neutral conditions. There was no statistically significant difference in the density, richness or EPT component of macroinvertebrate communities associated with iron precipitate substrate as opposed to substrate that has no precipitate.

However, it appears that there is some tendency for the sensitive Ephemeroptera, Plecoptera and Trichoptera to dominate the community associated with iron precipitate substrate after it has been placed in the circum-neutral stream for extended periods of time (4 weeks). This may be due to sensitivity to the toxicity of iron precipitate, or it is possibly a response to changes in the composition of the organic biofilm layer. At the commencement of the experiment all three types of treatments had equivalent biomasses of organic biofilm. However, the methods used did not allow for any qualitative analysis of the biofilm composition. It is possible that initially substrate with iron precipitate had epilithon layers consisting of micro-organisms such as ferromaganese depositing bacteria that are poor food source for grazing invertebrates (Wellnitz et al., 1994). After a period of time in Grasmere Stream these layers are probably replaced with diatoms and other periphyton that would be a more favourable food source, which in turn would be more attractive for sensitive invertebrate grazers.

Contaminated substrate such as iron-precipitate can affect the growth of macroinvertebrates (Day et al., 1995). Therefore, further investigation of AMD sediment and iron-precipitate could involve assessment of growth rates in macroinvertebrates, in addition to toxicology assays.

This investigation indicates that substrate in AMD catchments had increased biomass of organic biofilms in comparison to non-AMD impacted reference catchments. However, whether or not these biofilms are suitable as a food source for macroinvertebrate grazers is unclear.

Iron hydroxide precipitate that is present on the substrate in these catchments may be limiting the growth of epilithic micoorganisms, particularly photosynthetic taxa, the biomass of which may be replaced by blooms of ferrophilic bacteria.

It was not clearly apparent that macroinvertebrates actively avoid introduced substrate coated in iron oxide precipitate in a cicum-neutral environment, although the results of this investigation did indicate a marginally insignificant trend for sensitive taxa to be present in higher numbers after four weeks than after one week.

Chapter Five

Towards managed recovery of habitat and benthic communities in mine affected streams

Coal mining has been a significant part of the economy of the Buller District and indeed the entire West Coast region for more than 150 years. However, as other commercial enterprises are becoming increasingly important in the area, the importance of coal within the region is less prominent. One such industry is ecotourism, showcasing the natural attractions and assets of the region. With the advent of the Resource Management Act (1991), the value of natural ecosystems has become more economically tangible. Furthermore, a public environmental awareness and international expectation of better environmental practices places increased pressure on industries in the region to use environmentally sound procedures.

In order to optimise the tourism potential of the area, it has become apparent that the restoration of some of the many streams affected by historical mining activities is desirable to improve the ecosystem quality and the aesthetic appeal of the waterways.

Furthermore, the development of remediation and restoration techniques for impacted streams will result in an improvement in the negative environmental effects of existing mining operations. Although increased mining operations may be undesirable from an environmental perspective, the economic significance for the community of mining cannot be overlooked. Therefore, the establishment of effective remediation and restoration practices provides an effective foundation upon which a compromise between economic and environmental concerns can be offered.

The unique West Coast environment offers a challenge for effective remediation. For example, on the Denniston and Stockton Plateaus, the topography of affected streambeds is frequently very steep. There is little topsoil and many areas of exposed bedrock. This increases costs and logistical difficulties of the installation of remediation systems like wetlands. Furthermore, the nature of the soil in combination with the harsh climate creates difficulties for the re-establishment of riparian vegetation, an important component of stream restoration (Davis et al., 1997).

Many different techniques have been employed internationally for improving the quality of AMD impacted streams by increasing the pH and removing metals from AMD water. For example, the use of anoxic limestone drains to increase pH and remove metals from acid mine drainages entering streams (Cravotta & Trahan, 1999). Such systems are often intensive and require long-term management and financial commitment.

Passive systems are often developed that incorporate a variety of remediation techniques. These may include any of, or a combination of, ponds, wetlands, drains, limestone and organic matter (Skousen, 2001). All methods have both logistical and financial disadvantages. For example, in addition to the expense of purchasing and placing limestone, the use of calcium carbonate to increase pH can be problematic as it also causes metals to precipitate. This clogs the limestone and prevents it further neutralising AMD water (Skousen, 2001). Furthermore, with the use of biological processes to mitigate the effects of AMD comes a large financial outlay to create an artificial wetland environment, in addition to the logistical issues of identifying an appropriate site and the construction of it.

AMD remediation has traditionally focused on acidity as the chief cause of toxicity and degraded benthic macroinvertebrate communities (Robbins et al., 1999). Results of this study corroborate that low pH is the primary cause of toxicity to macroinvertebrate taxa (refer Chapter 3). However, some toxicity may also be attributed to the presence of dissolved metals. Furthermore, metals can also cause additional impacts such as precipitates (refer Chapters 3 and 4). A successful remediation method will be required to remove excessive concentrations of metals in addition to increasing the ambient pH of AMD water. If only pH is addressed, the remaining metals may have consequences for ecosystems, particularly as these can create toxic effects whenever metal-impregnated sediment is disturbed (Filion & Morin, 2000; Finlayson et al., 2000; Fajtl et al., 2002).

Metal pollution can profoundly influence freshwater invertebrate populations. This can affect the food web, and lead to changes in community composition, and biological interactions (Clements et al., 2000). Caddisflies caged in an AMD impacted stream had poor survival rates, and had high whole-body metal concentrations after just five days of exposure (DeNicola & Stapleton, 2002).

Sub-lethal metal exposure can cause invertebrates to rapidly alter their behaviour, and therefore alter the outcome of community interactions such as competition and predation. For example, Hydropsychidae caddis larvae become less able to compete for food resources, and therefore feeding declines as a result of other species assimilating limited food resources more efficiently (Vuori, 1994). Furthermore, many invertebrates alter their behaviour in response to predation, which may render some sensitive prey taxa more vulnerable than others (Kiffney, 1996; Clements, 1999; Lefcort et al., 1999).

Invertebrate communities in AMD contaminated streams are characterised by low species richness and abundance, and are dominated by Chironomidae (Winterbourn et al., 2000). In addition to the toxicity of low pH, metal contamination from mine drainages into streams changes the proportions of taxonomic groups within the communities. Dominant groups shift from generally being Ephemeroptera, Plecoptera and Tricoptera, to more pollution tolerant-groups such as Chironomidae, which can alter their life-histories in response to metal contamination (Groenendijk et al., 1999).

Improving water quality, both by increasing pH and removing metals, will improve both the abiotic and biotic environment in a stream (DeNicola & Stapleton, 2002). Therefore, by employing remediation systems that address both low levels of pH and elevated metal concentrations, managers can avoid many of the problems associated with treatments like clogging precipitates and prolonged habitat toxicity. Ultimately, this will lead to the entire stream habitat and environment improving, in addition to ameliorating the water quality. This will therefore optimise the ability of macroinvertebrates to recolonise impacted streams and establish persistent populations.

The removal of metals in addition to increasing pH may be particularly important for remediation of AMD impacted streams on the West Coast of New Zealand. New Zealand benthic macroinvertebrates lack specialisation and seem to be adapted to abroad range of conditions, rather than being specialised for certain environments (Winterbourn et al., 1981).

Studies in the United States have shown that mayflies sourced from rivers that are impacted by AMD have a better tolerance to elevated metals under experimental conditions (Clements, 1999). This suggests that there is an element of adaptation occurring in areas of moderate or minor AMD pollution. Streams that are acutely polluted are unlikely to have such sensitive taxa present. In contrast, studies with West Coast macroinvertebrates suggest that exposure to streams with naturally low pH, does not facilitate adaptation in benthic macroinvertebrates to AMD conditions (Chapter 3 of this study). This may be due to the general life-history strategies of New Zealand benthic invertebrates. A combination of the frequent high flow events of West Coast streams and the lack of habitat specialisation possibly results in the opportunistic colonisation of such streams. Therefore, animals sourced from West Coast streams for such experimentation may not necessarily have been present in these environments for generations, preventing the occurrence of adaptation.

In addition, the natural conditions in these waterways needs consideration. The high frequency of flood events in the area may render many conventional remediation methods ineffective. Furthermore, the naturally low background pH of groundwater and surface water in the area needs to be considered. For example, the background pH of surface waters on the Denniston Plateau is 4.5. The artificial addition of substances to increase pH would have to be moderated to ensure that pH is not raised to a level that is significantly higher than 4.5. An increase in pH above background levels may alter the natural chemical and biological processes, and have unforeseen impacts, particularly on downstream habitats. Furthermore, the source of invertebrate colonists would need to be near to the remediated stream. Therefore, a close approximation of their accustomed habitat and water chemistry is likely to optimise colonisation potential.

Rapid Creek is being used as a model catchment for testing remediation techniques. Several methods of mitigating the effects of mine impacted water are being trialled.

Several commercial products for AMD remediation are also available. While these have been successful in some places, for example Australian opencast tailings ponds (D.McConchie, pers comm.), and for remediating contaminated soils (Ciccu et al., 2003), they are untrialled with New Zealand freshwater macroinvertebrates.

The aims of this investigation were:

1. To test what level of dilution of AMD water results in improved invertebrate survival.
2. To establish if commercial AMD remediation products (Zeolite and Bauxsol™) improve mayfly survival.

METHODS

Experiments over 96-hour were used to test the ability of possible mitigation techniques to reduce AMD toxicity.

Experiment 5.1 Dilution of AMD as a remediation method

AMD water from Rapid Creek (downstream of Sullivan's Mine) was diluted with distilled water between 2 and 8-fold (Table 5.1).

Each treatment was replicated five times. Each replicate consisted of a single covered plastic acid-washed petri dish, containing 5-10 animals, and 40 mls of water. Petri dishes were continually aerated, and had additional water of the same treatment added to maintain 40 mls (Figure 3.1).

Mortality of the common mayfly, *Deleatidium* was measured as the response variable for these experiments. *Deleatidium* was selected as a study organism as it is easily maintained in a captive environment, and it is an abundant and easily collected genus.

This experiment was conducted in a controlled environment with an ambient temperature of 4.5°C, and a day/night light regime of 12:12 hours.

Deleatidium were assigned to dishes in a randomized block design to allow for statistical compensation of the variable sizes of study organisms.

Water from Rapid Creek, collected downstream of the Sullivan Mine adits, was diluted with distilled water in a series of doubling dilution. Distilled water was used as a control. Specific treatment details are outlined in Table 5.1.

Table 5.1 Treatments used in the AMD dilution experiment.

Dilution	Water type	Dilution ratio AMD:Distilled Water
AMD (no dilution)	Rapid Creek	1:0
2-fold	Rapid Creek	1:1
4-fold	Rapid Creek	1:3
8-fold	Rapid Creek	1:7
DW	Distilled Water	0:1

Mortality was monitored every 24 hours for four days. Individuals were gently prodded with a blunt probe, and if they failed to exhibit a response they were considered dead.

The pH and conductivity of each treatment was measured using a portable Oaktron® Waterproof pH/Con 10 Meter.

Statistical Analysis

Mayfly survival data collected at 96 hours were arc-sine/square-root transformed to normalise the data, and then analysed using a generalised linear model (GLM). An LSD post-hoc test was used to identify treatments that produced significantly different results. The SAS System for Windows, Version 8 was used for this analysis.

Experiment 5.2 Testing commercial AMD remediation technologies

Two commercial remediation products, Bauxsol™ and Zeolite, were tested using a 96-hour toxicity experiment. Specific treatments are detailed in Table 5.2.

Mortality of three common mayfly genera, *Deleatidium, Nesameletus* and *Coloburiscus* was measured as the response variable for these experiments. A randomized block design was employed for this experiment. Each treatment consisted of one replicate from each of the five blocks. Each unit of a block consisted of a standardized combination of mayfly sizes and genera. Overall mayfly survival was measured. Specific mortality of each genus was not assessed by this experiment.

This experiment was conducted in a controlled environment with an ambient temperature of 10°C, and a day/night light regime of 12:12 hours.

Water from Rapid Stream, just downstream of the Sullivan's Mine adits were treated with Bauxsol™ and Zeolite as described below.

Zeolite was packed into a glass cylinder, and control water (from Rapid Creek upstream of mine drainage), and AMD water (from Rapid Creek, immediately downstream of mine drainage) was slowly dripped through (a rate of 1 litre/12 hours) and collected (Figure 5.1). This 'remediated' water was then used for Zeolite treatments.

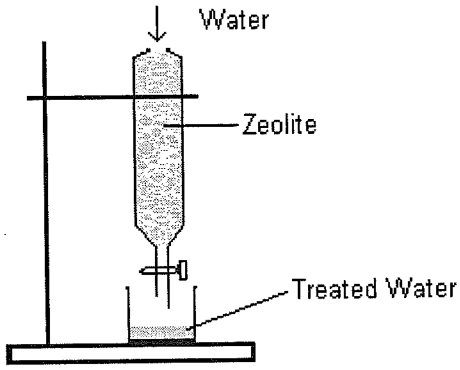


Figure 5.1 **Treating AMD water with zeolite. The tap allowed water to be dripped through very slowly, facilitating cation exchange and effectively reducing metals and raising the pH of Rapid Stream AMD water.**

Bauxsol™ was mixed into AMD water (1L) in a large beaker using a magnetic stirrer. Bauxsol™ was added in small increments until the pH was 8.3, according to the manufacturers' specifications. This was to insure the removal of zinc ions. The resulting pH of water treated by Bauxsol™ was considerably higher than background levels of the Rapid Creek catchment (pH 4.5). To mimic the natural water, the treated water was therefore adjusted post-treatment by the addition of 1M hydrochloric acid. The final product was filtered through Grade 1 filter paper to remove the fine Bauxsol™ particles.

Both Bauxsol™ and Zeolite were also used to treat non-AMD impacted water from the Rapid Creek catchment, in an effort to detect any negative effects of the treatments.

Mortality was monitored every 24 hours for four days. Individuals were gently prodded with a blunt probe, and if they failed to exhibit a response they were considered dead.

The pH of each treatment was measured using a portable Oaktron® Waterproof pH/Con 10 Meter before the commencement of the experiment.

Table 5.2 Treatments used for the assessment of commercial AMD remediation products.

Treatment Code	Origin of Water	AMD	Remediation Product
U-C	Rapid Creek, upstream of Sullivans Mine	No	None – control
D-C	Rapid Creek, downstream of Sullivans Mine	Yes	None – control
U-B	Rapid Creek, upstream of Sullivans Mine	No	Bauxsol™
D-B	Rapid Creek, downstream of Sullivans Mine	Yes	Bauxsol™
U-Z	Rapid Creek, upstream of Sullivans Mine	No	Zeolite
D-Z	Rapid Creek, downstream of Sullivans Mine	Yes	Zeolite

Statistical Analysis

The 96 hour survival of mayflies in each treatment was arc-sine/square-root transformed to normalise the data, and then analysed using a Generalized Linear Model (GLM). If the model identified some treatments as significantly different these were identified with a LSD post-hoc test. The SAS System for Windows, Version 8, was used for this analysis.

RESULTS

The dilution of AMD water with distilled water resulted in an increase in pH (Figure 5.2). The pH was 2.79 in the undiluted AMD water. After an 8-fold dilution, pH had increased to 3.37. This was still well below the background levels of natural streams in the vicinity of Rapid Creek that are not affected by mining activities (pH 4.5).

A linear decrease in conductivity was also observed in diluted AMD water treatments (Figure 5.2). Conductivity was >1000 μScm^{-1} in undiluted AMD water, but dropped to <200 μScm^{-1} in the 8-fold diluted water treatment. Because the majority of the ions in AMD water causing the high conductivity are most likely dissolved metals like iron, aluminium zinc and nickel, it was assumed that this reduced conductivity should be due to the decrease in the concentration of dissolved metals.

Figure 5.3 illustrates mayfly survival in the treatments. In the distilled water treatment, 80% of individuals survived the 96-hour period, whereas less than 10% of individuals in the AMD dilution treatments survived. The GLM procedure suggested that one or more treatments was significantly different (Table 5.3), and the LSD test confirmed that this was the distilled water treatment.

Table 5.3 Results of the GLM procedure analysing differences in mayfly survival between the dilution treatments. NB * indicates statistical significance at the p<0.05 level.

	DF	Mean square	F	p
Block	4	0.049	2.62	0.07
Treatment	4	1.084	58.54	0.0001*
Error	16	0.019		
Total	24			

At 96 hours there was no significant difference in the survival of mayflies in AMD water that was untreated, or had been diluted 2, 4 or 8-fold.

At 72 hours, the three diluted treatments still had some mayfly survival, whereas the undiluted AMD water had 100% mortality. Mortality was greater than 95% after 96 hours in all AMD water treatments.

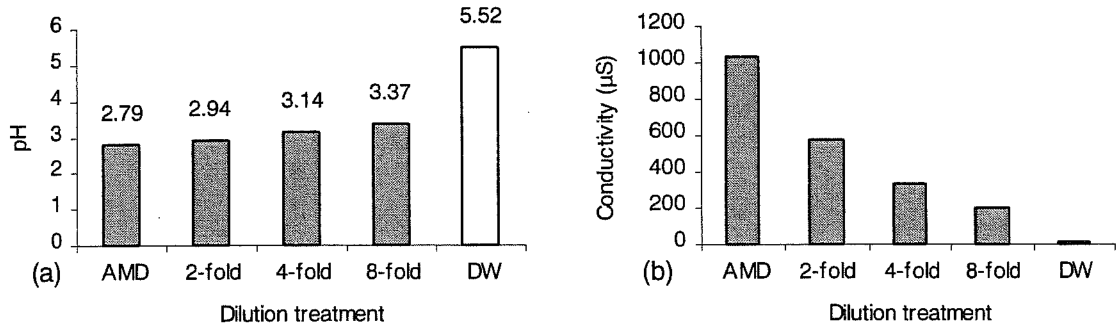


Figure 5.2 **pH (a) and conductivity (b) of AMD water after dilution with distilled water. DW is the distilled water control treatment. For full treatment explanations see Table 5.1.**

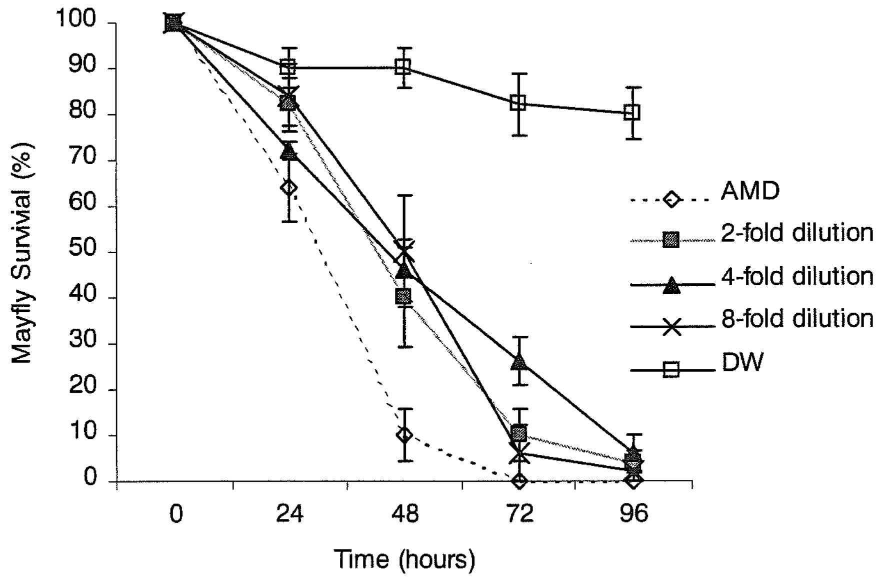


Figure 5.3 **Mean survival of mayflies (±SE) in diluted AMD water and distilled water (DW).**

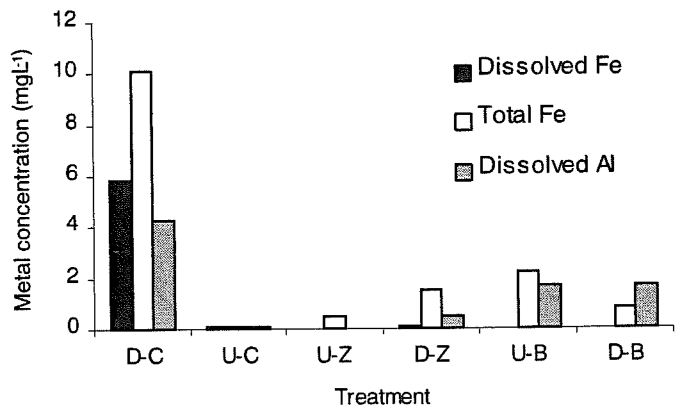


Figure 5.4 **Total iron, dissolved iron, and aluminium concentrations in treatments. D-C is untreated AMD water (from Rapid Creek), U-C is untreated non-AMD water, U-Z is non-AMD water treated with Zeolite, D-Z is AMD water treated with Zeolite, U-B is non-AMD water treated with Bauxsol™, and D-B is AMD water treated with Bauxsol™.**

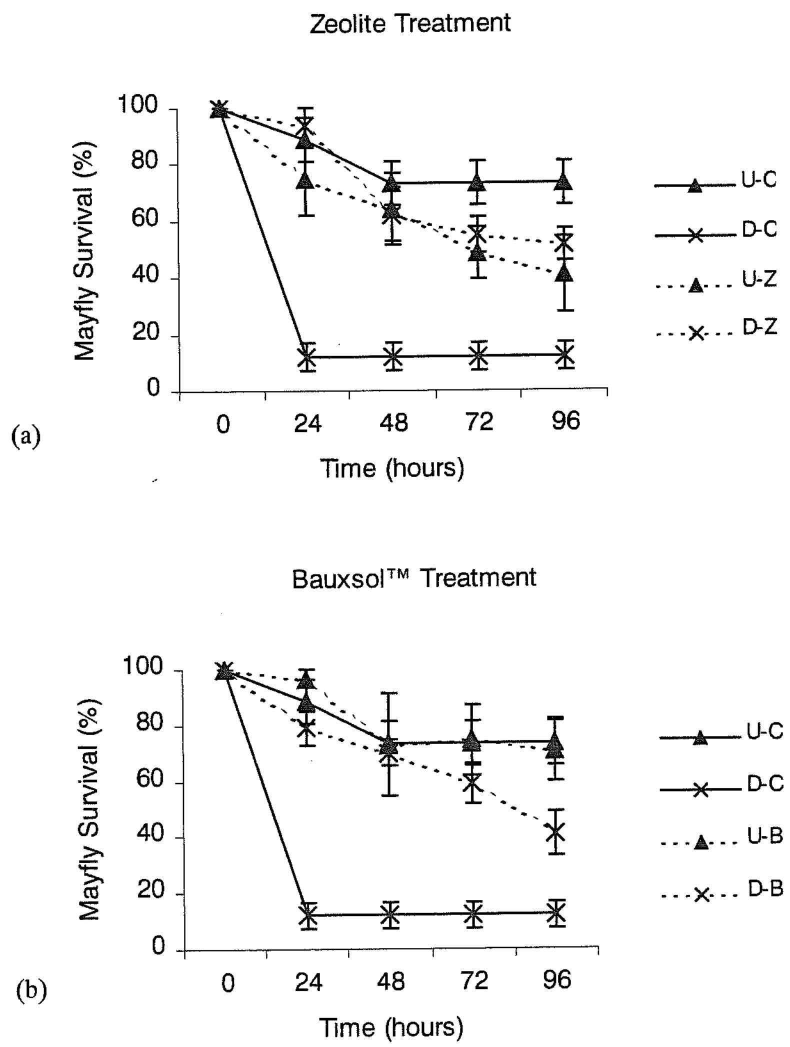


Figure 5.5 **Mayfly survival in AMD water and non-AMD water treated with Zeolite (a), and Bauxsol™ (b). Triangles denote non-AMD water; and crosses are AMD. Dotted lines denote treatments; solid lines are controls.**

Treating AMD water from Rapid Creek with Bauxsol™ and Zeolite reduced the concentration of total iron, dissolved iron and dissolved aluminium (Figure 5.4). Iron was effectively removed by both products, and dissolved iron in particular was almost totally removed from Rapid Creek AMD water when treated. Zeolite appears to remove a greater proportion of dissolved aluminium (>75%) than Bauxsol™ does.

Water from Rapid Creek that was upstream of the mine (non-AMD) had very low concentrations of iron and aluminium when used as a control. However, when this water was treated with Zeolite and Bauxsol™ the concentrations of these metals increased (Figure 5.4), suggesting that exposure of water to these products could be releasing a small amount of ions into solution.

Zeolite treatment increased the pH of Rapid Creek AMD water from 3.2 to 4.3. The pH of the water treated with Bauxsol™ was increased to 8.2 as per the manufacturers instructions for the removal of zinc ions, then re-adjusted to 4.5 with hydrochloric acid.

More mayflies survived when exposed to AMD water treated with either Bauxsol™ or Zeolite, than untreated AMD water. For the first 48 hours of the exposure period, mayflies appeared to survive equally well in treated AMD water as they did in non-AMD water (Figure 5.5). Only the untreated AMD water produced mean survival rates significantly lower than the others (Table 5.4). Survival in the treated AMD water was reduced at 72 hours, but after 96 hours the two commercial treatments were supporting mayfly survival rates that were 50% greater than untreated AMD water, and still significantly different (Table 5.4).

Table 5.4 **Results of the GLM procedure analysing the survival of mayflies in AMD water treated with Zeolite and Bauxsol™. NB * indicates significant at $p<0.05$ level.**

	DF	Mean square	F	p
Block	4	0.015	0.16	0.955
Treatment	5	0.449	4.74	0.005*
Error	20	0.095		
Total	29	4.202		

When control water (upstream of mine drainage) was treated with these products, Bauxsol™ did not cause additional mayfly mortality, whereas Zeolite caused significant mortality (Figure 5.5). The mean survival of mayflies in Zeolite controls was the same as that in Zeolite treated AMD water, whereas the Bauxsol™ treated control had the same mean survival as the untreated control.

DISCUSSION

The dilution of AMD contaminated Rapid Creek water does not appear to be an effective method of remediation. Dilution up to 8-fold did not significantly improve the survival of mayflies exposed to AMD water. It is likely that this level of dilution is inadequate to raise the pH to a less toxic level. By extrapolating along the curve of pH increase and dilution, it is predicted that AMD water from Rapid Creek will be of a pH that is tolerable by most benthic invertebrates (>4) if it has been diluted approximately 64-fold. This degree of dilution would also render the dissolved metal concentrations in the AMD water non-toxic to most taxa.

Although such levels of dilution may make Rapid Creek and other AMD impacted waterways more inhabitable for invertebrates, it is scarcely feasible to actively dilute the waterway to this level, as Rapid Creek has a stable flow of AMD water entering it from Sullivans Mine (A.Black, pers comm.). Therefore, in order to establish a dilution system, excessive amounts (64-fold dilution of the mine discharge) of 'clean water' would have to be diverted into Rapid Creek. The addition of such vast of quantities water would entirely alter the morphology and nature of Rapid Creek.

Dilution, therefore, is not considered a suitable remediation method for small AMD impacted streams such as Rapid Creek. It may however, be the cause of passive recovery occurring in larger catchments that have a number of undiluted tributaries joining the impacted channel (refer Chapter 2). For example, Charming Creek has two tributaries entering it every kilometre, which provide a high level of dilution.

The use of Zeolite appears to render Rapid Creek water less toxic to the common mayfly. The concentrations of dissolved metals are markedly reduced and the pH of the water is increased to levels that are tolerable by mayflies (>4). Treating Rapid Creek water that was not AMD impacted (sourced upstream of the mine) resulted in similar levels of survival to treated AMD water. These survival levels however, were not as high as those of animals exposed to upstream water that was not treated with Zeolite. This suggests that there may be some negative affects from the use of Zeolite and of over-treating Rapid Creek water. These negative affects may be partially due to the presence of a very fine residue that was observed in the Zeolite treatments. It appeared to be washed out of the Zeolite, and tended to settle in experimental containers. This may be less of a problem in-situ, as this fine dust would most likely be flushed through the system. However, Zeolite also appeared to increase the levels of some ions in the water. This may indicate some underlying chemical effect of Zeolite treatment. This is likely to be particularly important in the West Coast environment where periods of high rainfall may frequently lead to the dosage of Zeolite fluctuating. Therefore, if Zeolite was to be used in situ, it would need to be monitored to prevent excessive quantities being flushed into the water. This is further compounded by the process of treatment. AMD water needs to pass through the Zeolite granules very slowly in order to optimise the cation exchange process. Using Zeolite to mitigate the effects of AMD water would therefore need to be actively monitored and managed in order to ensure that it optimally increases the pH and removes dissolved metals, without adding an additional stressor to aquatic life.

Bauxsol™ also effectively removed metals from AMD Rapid Creek water. The survival of mayflies exposed to AMD water that had been treated with Bauxsol™ was significantly improved. Furthermore, there appeared to be no effect of 'over-treatment', as non-AMD water treated with Bauxsol™ appeared to have no additional lethal effects on mayflies. Survival in the upstream water treated with Bauxsol™ was as high as the control treatment of untreated non-AMD water. Because of this, Bauxsol™ possibly could be a more passively employed treatment method than Zeolite, as dosage would not have to be as closely monitored and adjusted according to flows. However, in order to ensure zinc ions were removed, Bauxsol™ had to be added in such quantities that raised the pH to 8.3. Therefore, Bauxsol™ would have to be used in conjunction with a system to return the pH of the water to level more analogous to that of the background levels of the area. In this study, this was achieved by adding hydrochloric acid, although this may not be a suitable procedure for re-acidifying significant volumes of treated water in the field.

Chapter Three of this study has suggested that pH appears to be the primary cause of AMD toxicity. Liming is commonly used overseas to address acidic pH, particularly in streams affected by acid precipitation (Herrman & Svensson, 1995). Liming has been shown to be an effective method of decreasing metal concentrations in some AMD impacted lakes and streams, in addition to increasing pH and conductivity. The addition of lime also decreases the transparency as both phytoplankton and zooplankton increase in abundance and diversity (Stenson et al., 1993), particularly in low velocity or standing water.

A disadvantage of liming is that it must be repeated frequently to maintain optimum pH levels. This is simple in streams as a dosing device can be installed to release lime in proportional to the discharge (Fleischer et al., 1993). Liming can however damage sensitive vegetation in wetlands, and some locally extinct species may have difficulties recolonizing lime-treated areas (Fleischer et al., 1993).

Anoxic limestone drains have also been used in AMD (Skousen, 2001). These drains can effectively neutralise effluents and produce pH levels greater than six. However, increasing pH can cause precipitates of minerals in high concentrations to form, and reduce the effectiveness of anoxic limestone drains by physically clogging them and reducing the surface area available for chemical reactions (pH neutralisation) (Robbins et al., 1999). However, limestone drains are still considered effective for neutralising mine effluent, despite becoming coated in iron and aluminium hydroxides (Cravotta & Trahan, 1999).

It is likely that any method of AMD remediation will need to be part of system, and will not be successful in isolation. Most successful remediation efforts involve the use of a wetland or settling area, which serve a variety of purposes. These include collecting precipitates and toxic sediments, and the microbiological processing of dissolved compounds in AMD water (Skousen, 2001). Chen, Soulsby and Younger (1999) found that a comibination of liming and passive aerobic wetlands was the most effective way of neutralising acid mine drainages.

The results presented here indicate that with respect to macroinvertebrates, increasing pH may be an important initial step for remediating AMD degraded streams. However, although the reduced pH of AMD in the Buller area seems to be the most significant cause of benthic invertebrate mortality, the apparently less significant toxic effects of dissolved metals may increase as pH is raised by remediation. Aluminium and iron ions in transition from acidic water to neutral can cause acute toxicity (Soucek et al., 2001). In addition to this acute toxicity, sub-lethal toxicity may also affect the food-web as metals may accumulate in body tissues and be transmitted through the trophic levels.

For example, aluminium can be highly toxic to many invertebrates as well as fish and plants. Aluminium tends to accumulate in the bodies of shredders and deposit feeders (Herrman & Frick, 1995). This may therefore lead this trophic level becoming a toxic food source for their predators. However, aluminium is not thought to be biomagnified in the food chain (Herrman et al., 1993; Winterbourn et al., 2000), although this may simply be due to the food-chain being truncated by toxicity.

Because metals have complex affects on macroinvertebrate stream communities both through toxic and chronic mechanisms, it is important that this aspect of water chemistry is considered in remediation efforts.

Some studies suggest that freshwater communities will recover from the effects of metal contamination if sufficient habitat remediation is achieved (Nelson & Roline, 1996; Prat et al., 1999; Wantanabe et al., 2000). However, this will rely on both the proximity of a healthy pool of colonists to replace taxa that have become locally extinct, and on the maintenance of habitat integrity.

Impacted streams in the Denniston area, including Rapid Creek are within reach of colonists from healthy streams. Often this is as simple as the upper reaches of the stream. For example, Rapid Creek has a diverse macroinvertebrate community, particularly insects, upstream of the Sullivan Mine adits that will enable re-colonization of impacted reaches both by drift and aerial colonization by adults.

This proximity of colonists suggests that AMD impacted streams like Rapid Creek may be able to be rehabilitated, provided the water chemistry is adequately remediated. The commercial products Bauxsol™ and Zeolite seem to do this, although when used in the West Coast environs such products would need to be a component in a mitigation system, rather than as stand-alone methods.

Increasing the pH and removing the high concentrations of metals in streams such as Rapid Creek will lead to a gradual increase in the water quality, which in turn should lead to a partial recovery of the aquatic invertebrate community and an improvement in the aesthetic appeal of such streams.

Chapter Six

Conceptually modelling the recovery of AMD impacted streams: natural recovery, remediation systems and recommendations for management

Current and historical coal mining on the West Coast of the South Island has resulted in a number of streams being affected by acid mine drainage (AMD). The development of a concept based understanding of the processes occurring in these ecosystems would be beneficial when considering management strategies.

Streams can be affected by AMD to varying degrees, depending on the magnitude of initial and ongoing impact, as well as the specific geographical, physical and chemical conditions in the receiving stream. Because of this the ecosystems of some affected streams show a level of natural recovery without anthropogenic intervention. Conversely, the condition of severely impacted streams, such as Rapid Creek, indicate that without active management little of no recovery of the stream community will occur.

Stream length can be a useful mitigating factor for AMD impacted stream systems, as this increases the likelihood of unpolluted tributaries entering the system. Unimpacted tributaries dilute metal contaminants and hydrogen ions, allowing an increase in pH and decrease in metal toxicity to sensitive macroinvertebrate taxa. Furthermore, longer catchments may also provide buffering for contaminated stream water being buffered with varying streambed geology types, and changes in vegetation or land use types.

A conceptual model of the response of the benthic macroinvertebrate community was constructed to illustrate this relationship (Figure 6.1). As macroinvertebrates are generally considered good indicators of overall ecosystem health (Nelson & Roline, 1999), it can be considered that this reflects a typical ecosystem of an AMD impacted waterway.

Figure 6.1 indicates that a degree of natural recover from AMD contamination can occur in impacted streams. However, this is directly associated with the dilution potential of tributaries, and indirectly associated with the size of the catchment, and the length of the stream, as this increases the probability of such tributaries existing.

Ecosystem recovery is likely to be signalled in the benthic community by an increase in diversity and shift in community composition. The longitudinal survey comparing AMD catchments with unimpacted reference catchments suggests that an 'optimum' community would likely consist of 15-20 taxa, as opposed to AMD impacted communities, which generally have less than 10 taxa (Chapter Two). Furthermore, optimum communities, such as those found in the Little Wanganui River, Tidal Creek and Chasm Creek, are dominated by Ephemeroptera, Plecoptera and Trichoptera taxa, rather than Diptera as in AMD impacted catchments (Chapter Two). EPT are considered a useful indicator of metal contamination due to their sensitivity, as opposed to the relative tolerance of some Diptera taxa (Hickey & Clements, 1998).

Although such recovery may occur, it is unlikely that an optimum macroinvertebrate community will establish in impacted streams. Communities may recover some diversity and acid or metal sensitive species may occur in sites that have significant longitudinal distance from the source of AMD. However, in general, the truncated nature of West Coast stream systems, particularly in the Denniston/Stockton area, and the magnitude of AMD contamination generally results in the water chemistry of impacted streams not recovering sufficiently to support such communities.

Like many streams on the West Coast, the streams of the Denniston Plateau are naturally acidic (approximately pH 4.5). Macroinvertebrate communities in these streams are able to tolerate the low pH, provided it is 4.5 or greater (Collier, 1988). However, the markedly low pH in AMD impacted streams (often less than pH 3.5) is a significant mechanism in excluding many sensitive taxa (Chapter Three) and generally degrading the condition of the biotic community.

Figure 6.2 illustrates the concept that the survival of sensitive taxa improves significantly as pH is increased, without any further action to remove toxic dissolved metals. Experimentation indicates that this improvement seems to plateau at a pH of 4 (Chapter Three), which although still low, seems to be tolerable by many taxa in the vicinity of AMD contaminated streams on the Denniston Plateau. Therefore, with such low-pH tolerant taxa as a source of colonists, a moderate increase in pH in AMD impacted streams may facilitate an amelioration of the benthic community, and in turn ecosystem function.

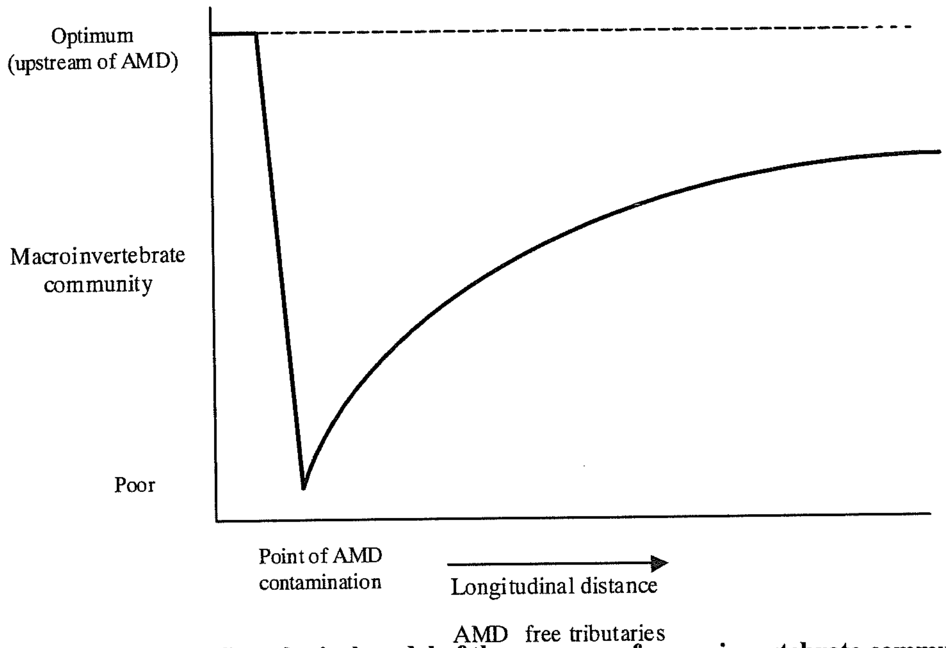


Figure 6.1 **Hypothetical model of the response of macroinvertebrate communities in AMD impacted streams in the Buller District as longitudinal distance/number of AMD-free tributaries entering the mainstem increases.**

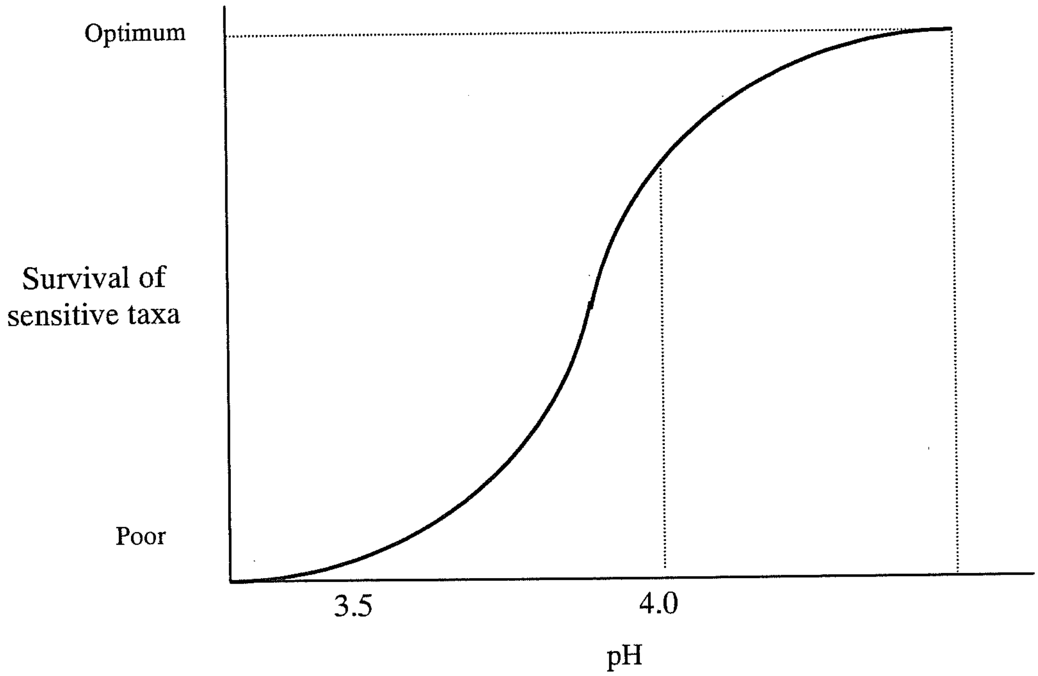


Figure 6.2 **Conceptual model of the association between pH and the survival of sensitive taxa in streams on the Denniston Plateau.**

Many small AMD impacted streams in the Buller District have very low levels of pH, often less than 3. It is expected that natural dilution can increase pH to levels tolerable by sensitive invertebrates. However, in catchments such as Rapid Creek it appears that very high levels of dilution would be necessary to increase the pH sufficiently. A hypothetical extrapolation from data collected in this study (Chapter 5) suggests that in a situation such as Rapid Creek, where the pH of the main-stem is generally less than 3, dilution would probably need to be more than 64-fold to increase the ambient pH to 4 (Figure 6.3). Truncated streams like Rapid Creek are therefore unlikely to show a recovered macroinvertebrate community through improved water quality due to dilution. Longer catchments are more likely to have tributaries entering them, providing a level of dilution. Furthermore, the initial magnitude of AMD impact is likely to be lower in a larger river that has higher buffering capacity.

Active remediation of AMD contaminated streams like Rapid Creek using the commercial products such as Bauxsol™ and Zeolite should improve the water quality to some degree. However, sensitive taxa like mayflies still suffer mortality in AMD water that has been treated with these products (Figure 6.4). Therefore, it is expected that only a partial improvement in the macroinvertebrate community with the use of these products will occur, as some sensitive taxa may still be excluded by residual toxicity or limited by secondary effects of remediation products.

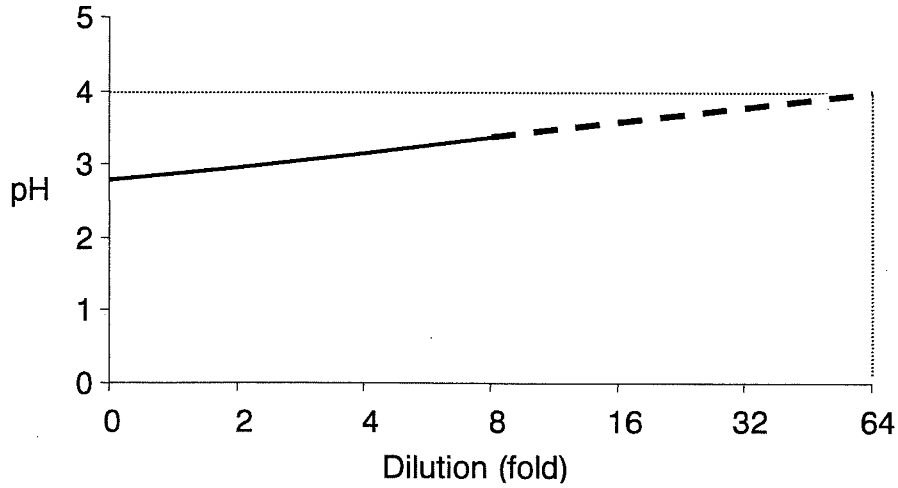


Figure 6.3 **Extrapolation of the increase in the pH of AMD impacted Rapid Creek water with dilution. Extrapolation is indicated by a broken line.**

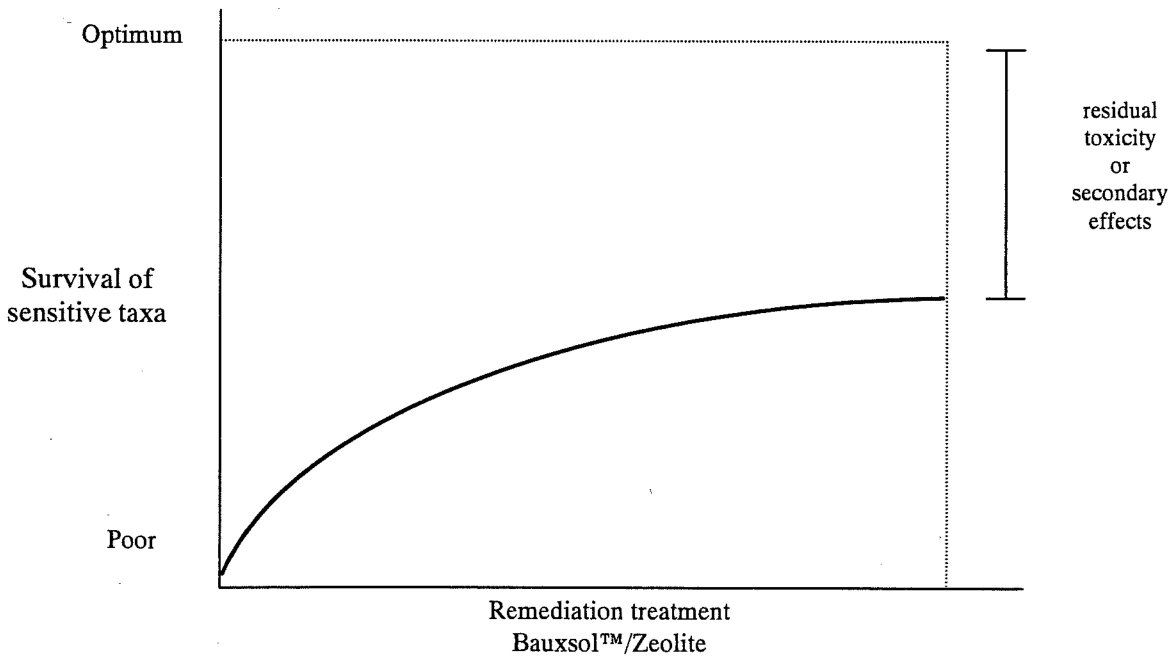


Figure 6.4 **Conceptual model of the response of sensitive taxa to an AMD impacted stream treated with commercial remediation products to remove toxic metals and increase pH.**

By considering all the aspects of AMD impacted stream communities, the response of specific taxa to various sets of conditions and types of treatments, I propose a conceptual model of recovery (Figure 6.5). Although grossly simplified, this model indicates the processes a stream ecosystem would experience leading to the recovery or establishment of a macroinvertebrate community analogous to those of non-impacted streams in the vicinity.

Like any ecosystem, a stream community recovering from AMD would be affected by numerous variables and stochastic events. This model attempts to incorporate some of these in an equation for the exponential part of the curve.

The gradient of improvement in 'macroinvertebrate health' can be represented by the equation:

$$y = a^{x}$$

Where:

y = the rate of recovery of the macroinvertebrate community

a = the effectiveness of remediation (or the inverse of the magnitude of impact)

x = the proximity of a source of colonists for the recovering community.

The physical stream might undergo several phases as the macroinvertebrate community recovers. Initially, an improvement in water chemistry would reduce or inhibit the deposition of iron precipitates. With natural weathering and abrasion during high flow events, iron precipitate would be removed from the substrate and the quality of the benthic habitat would improve.

The reduction in iron-precipitate deposition would facilitate the establishment of a periphyton community more comparable to that of healthy streams in the vicinity, as iron oxides negatively affect periphyton production and diversity (McKnight & Feder, 1984; Niyogi et al, 1999). It is expected that this periphyton, in combination with a recovering macroinvertebrate community, would represent improved ecosystem function.

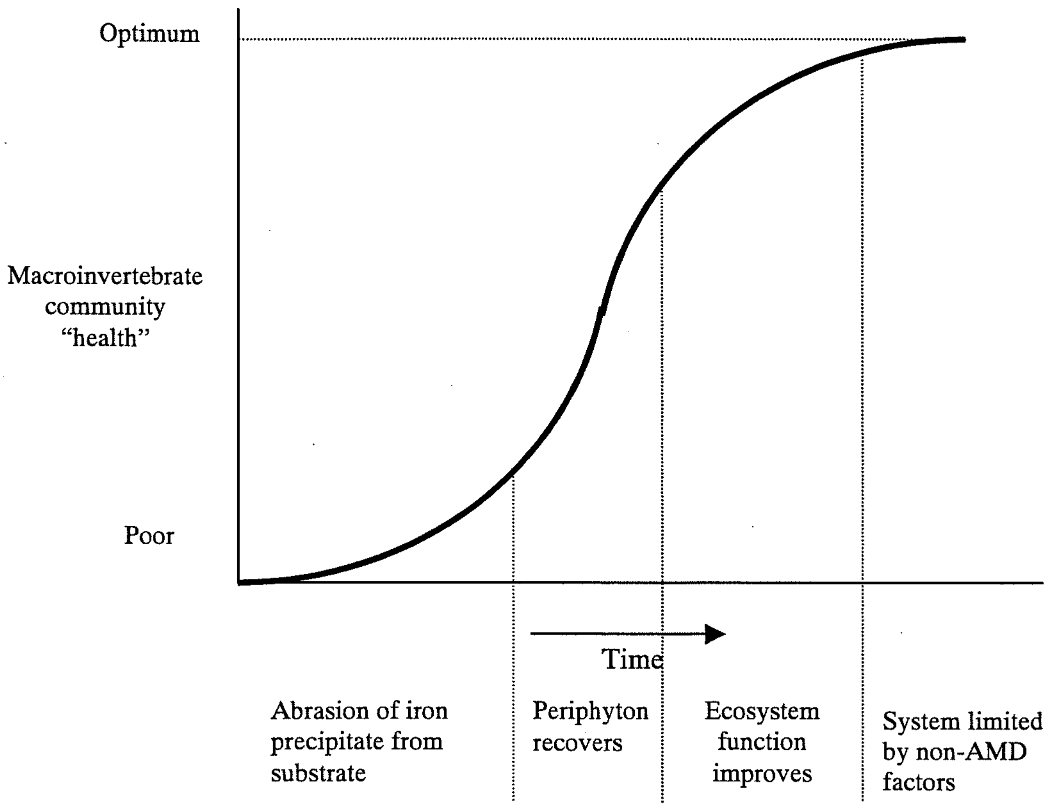


Figure 6.5 **Conceptual model of factors associated with the recover of AMD impacted streams through time**

The effectiveness of remediation (constant 'a') is likely to be influenced by several factors. For example, the number and nature of sources of AMD is very important. Remediation or recovery is likely to be more tangible where the point of contamination is a single, above ground discharge, such as the introduction of AMD to Charming Creek via the single source (Wearne Creek), as opposed to situations like that of the Miller Creek system where AMD is leaching into the stream water through numerous ground water seeps, which are more difficult to identify and address (Chapter Two).

This is related to another factor that will influence the effectiveness of remediation: the severity of the AMD source. Mine drainage varies in its water chemistry characteristics and receiving streams vary in buffering potential, so therefore the extent to which a benthic stream community is influenced is specific to each AMD source. For example, the macroinvertebrate community of the Waimangaroa River appears to recover partially, whereas that of Miller Creek does not. This may be at least partially associated with the fact that AMD impacted reaches of the Waimangaroa River were less severely impacted (pH levels >4.5, conductivity <100 μScm^{-1}) than those of Miller Creek, which had a pH less than 3, and conductivity more than 1000 μScm^{-1} (Chapter Two).

A further factor affecting the degree of recovery in an AMD impacted stream is the proximity to the source of contamination. As longer catchments such as the Waimangaroa River and Darcy Stream have shown an element of natural recovery, it seems that longitudinal distance is important if site-specific macroinvertebrate community health is considered. It is less significant when considering the entire stream length as an ecosystem.

Similarly, the temporal proximity of to the cause of AMD contamination is also likely to influence the ability of a stream macroinvertebrate community to recover (Wantanabe et al., 2000). Water quality may gradually improve after the termination of mining activities, or long term of contamination may cause adaptation in macroinvertebrate populations to allow them to tolerate the conditions.

Although based on the findings of this study, this model is essentially hypothetical. It also incorporates many assumptions. The most significant of these is that it assumes that the AMD remediation is constant. Any lapses or contamination events would probably remove colonists and effectively reset the system (Merrett et al, 1991).

Future research could beneficially test this model in stream ecosystems in the Buller District, and examine its applicability to AMD impacted systems in other areas.

This model predicts that with sufficient improvement in water chemistry the macroinvertebrate community would increase in density and diversity, generally to be considered 'healthier'. This in turn may lead to improved ecosystem function, facilitate colonisation by fish taxa, and improve the riparian vegetation of the stream. However, it is important to note that the curve of the model reaches a plateau. This signifies that regardless of remediation effort, limiting factors will always restrict a stream ecosystem. In the Buller District, particularly in the Denniston area, these factors include a naturally low pH, harsh climate, and lack of streambed stability.

Recommendations for management of AMD impacted streams in the Stockton/Denniston area

Many factors affect stream communities, and therefore when addressing a specific contamination issue such as acid mine drainage, a holistic approach to remediation is optimal.

The water chemistry of AMD inputs into streams is the cause of reduced macroinvertebrate community health. While several aspects of typical AMD water chemistry have adverse effects on macroinvertebrates and stream habitat, this study indicates that pH is the most important factor. Therefore, addressing the issue of low pH is likely to be the most useful issue for managers to concentrate on initially.

However, in order to ensure a holistic approach, and enhance the success of remediation efforts, a combination of restoration activities/remediation techniques is recommended. For example, while the addition of limestone products to AMD impacted streams may effectively increase pH, a successful long term strategy would be to include the construction of a wetland system to take advantage of biological processes that can moderate pH and remove dissolved metals (Skousen, 2001; Warren & Haack, 2001).

Furthermore, as the condition of stream systems is directly associated with the condition of the surrounding catchment (Baron et al., 2002), restoration is most effective when attempted at a catchment scale (Kauffman et al., 1997). Therefore, attention to the surroundings of a degraded stream is likely to be beneficial when attempting remediation. Coal mining, both by opencast and underground methods, can alter the surrounding landscape considerably. For example, open cast mining produces waste rock stacks and pits, and underground mining can lead to slumping. As a result, the degraded landscape is often denuded of vegetation and topsoil. However, active restoration with native plants has lead to greater than 40% of affected areas having ground cover within five years (Davis et al., 1997). Riparian vegetation along impacted streams would have the expected benefits of adding shade and cover, stabilising stream banks and stream bed, therefore increasing the accrual of periphyton resources (Friberg et al., 1997). Furthermore, riparian vegetation adds allochthonous organic matter to the water, which is an important input of energy in small streams (Fisher & Likens, 1972). In addition to these benefits, riparian vegetation would counteract some of the negative effects of active mining operations. For example, vegetation may reduce the amount of fine particulate matter, such as coal fines or roading material, entering the stream through run-off, and moderate the entry of acid mine drainage into the stream.

A successful management strategy would also include regular monitoring of remediation operations and stream macroinvertebrate communities. As any lapse in the treatment of AMD water would severely impede the recovery of the community, systems would need to be robust and well maintained.

Finally, all issues associated with AMD contamination should be considered before any future mining operations commence. With additional considerations for the affect of this phenomenon, many of the problems caused by acid mine drainage may be avoidable at the planning stage.

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